Comparative Risk Assessment and Environmental Decision Making

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Comparative Risk Assessment and Environmental Decision Making

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PREFACE

Decision-making in environmental projects is typically a complex and confusing process characterized by trade-offs between socio-political, environmental, and economic impacts. In many situations, decision makers have little incentive or ability to consider scientific assessments of project alternatives, and therefore select options that are promoted by the most influential stakeholders or politicians, thus disregarding the less dominant stakeholders and sometimes even degrading the environment. A framework is therefore needed that integrates risk assessment and engineering options; generates performance standards; compares options for risk reduction; communicates uncertainty; and effectively allows integration of stakeholder opinions in the decision-making process.

The idea for this book was conceived at the NATO Advanced Research Workshop (ARW) on "Assessment and Management of Environmental Risks: Cost-efficient Methods and Applications[1]" (Lisbon, Portugal, October 2000). The goal of the Lisbon workshop was to present risk assessment as a unified technique for providing a scientific basis for environmentally sound and cost-efficient policies, strategies, and solutions for various environmental problems. One of the workshop suggestions was to organize a more focused topical meeting on the application of specific risk-based techniques in developing Mediterranean countries.

The NATO Advanced Research Workshop in Italy was an important step in the development and application of comparative risk assessment (CRA) and other risk-based decision-analytical tools in environmental management. Comparative Risk Assessment (CRA) is a methodology applied to facilitate decision-making when various activities compete for limited resources. CRA has become an increasingly accepted research tool and has helped to characterize environmental profiles and priorities on the regional and national level. CRA may be considered as part of the more general but as yet quite academic field of multi criteria decision analysis (MCDA). Considerable research in the area of MCDA has made available methods for applying scientific decision theoretical approaches to multi-criteria problems, but its applications, especially in environmental areas, are still limited.

The papers presented in this book discuss issues ranging from specific and local studies (specific site, ecosystem, pollutant) to global decision and management frameworks (watersheds, regions, integration of multiple pollutants and stressors); they develop a range of approaches starting from specific methods to widely applied public policies. The papers show that the use of comparative risk assessment can provide the scientific basis for environmentally sound and cost-efficient policies, strategies, and solutions to our environmental challenges.

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[1] Linkov, I., Palma Oliveira, J.M., eds "Assessment and Management of Environmental Risks," Kluewer, Amsterdam 2001.

The organization of the book reflects sessions and discussions during the workshop. The papers in the introductory Chapter review and summarize developments in the fields of CRA and MCDA. They provide the necessary theoretical foundation as well as examples of applying these tools in environmental settings. Two topical chapters of the book summarize the results of discussions in working groups and plenary sessions. Each chapter reviews achievements, identifies gaps in current knowledge, and suggests priorities for future research in topical areas. Group reports written by group chairs and rapporteurs present a number of consensus principles and initiatives that were suggested during the group discussions.

The third chapter illustrates the application of risk and environmental assessments in different countries. Many papers in this chapter cannot be classified strictly as risk assessments, but they present the interpretation and perception of risks by individual scientists as well as illustrate a wide variety of environmental problems in developing countries.

An important objective of the workshop was to identify specific initiatives that could be developed by those in attendance and their broader network of institutions to enhance the progress of environmental risk assessment in developing countries. ARW participants represented a variety of organizations, from government agencies, industry, and research institutes to private consulting firms and academia. This group jointly proposed a number of specific recommendations relating to more effectively developing, using, and sharing information – including environmental data, assessment methods, and results. Highlights of these recommendations are presented in the working group reports in this book.

The workshop received high approval ratings from participants, with many commenting on the excellent opportunities for discussion. Instead of following the standard format of plenary and technical sessions dominated by speaker presentations, which typically allows little time for group synthesis, the agenda and activities were organized to foster interaction. Although 19 countries were represented, the number of participants was relatively small and allowed fruitful discussions throughout, from the working groups to the joint exercise and panel-facilitated topical segments. The meeting gave participants new insights and contacts, and many formal and informal collaborations were established.

Igor Linkov and Abou Bakr Ramadan
October 2003.

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Introduction

COMPARATIVE RISK ASSESSMENT: PAST EXPERIENCE, CURRENT TRENDS AND FUTURE DIRECTIONS

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Abstract

Comparative risk assessment (CRA) has become an increasingly accepted research tool and has helped to characterize environmental profiles and priorities on the regional and national level. Micro studies in comparative risk assessment are comparisons of interrelated risks involved in a specific policy choice (e.g. drinking water safety: chemical versus microbial disease risks) and focus often on one or a few types of environmental problem. At a larger scale are statewide or nationwide applications of CRA or programmatic CRA that consider multi-risks facing the society by comparing different types of environmental problems. This chapter reviews micro studies and macro applications of comparative risk assessment and then discusses future directions.

1. Comparing Risks

Risk comparison is a task at least as old as human thought. Should we drink dirty water or go thirsty? Should we eat an unfamiliar mushroom or hunt a large horned animal or go hungry? The formal analytics of risk date back about one century, with most of the important advances taking place within the past 50 years. It is only since the late 1980s that the comparative risk paradigm has entered widespread use to inform environmental decision-making. It is now commonly used to help understand the relative impacts of different environmental threats: Which is worse air pollution or water pollution, the depletion of stratospheric ozone or the increase in ground-level ozone? Skin cancer from sunlight or health problems from dietary pesticide intake?

The answers to such questions determine the action we take. Thus it is meaningful to analyze how we typically determine which action to take. As individuals, before we take action on anything, we predict the outcome of our action, weight it

I. Linkov and A. Bakr Ramadan (eds.),
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against other options and then decide which action to take, possibly in what order. This type of approach allows us to convince ourselves that we picked the best action that suits our needs. Thus, it is natural for institutions to follow a similar process. However, different options for actions, and prediction and evaluation of the consequences of each action of an institution may not be straightforward due to conflicting needs of a variety of stakeholders. Thus a more structured methodology is required for such more complicated cases. The decision tool that formalizes this type of approach for taking action is comparative risk assessment, which simply stated is an extension of this general behavior where one considers possible options before making decisions. The consequences have traditionally been analyzed in terms of risks, which indicates that the analysis is simply a *prediction* of the outcome. The word risk also suggests the probability of an *undesirable* outcome in accordance with typical applications of comparative risk assessment. However, we suggest that CRA should be considered in a broader way, as a way of thinking about acting rather than a focused tool to determine the least undesirable of all possible actions. As discussed in greater detail in the paper, this type of broader context leads to a more balanced tool with respect to which parties' views it represents. Thus, in its most general context, comparative risk assessment is a "mind-opening" approach for analyzing the outcomes of different scenarios to be able to make the most appropriate risk decision that will lead to a particular action.

2. Risk Perspectives and Comparative Risk Assessment

At its simplest, risk analysis involves multiplying a probability times a consequence to yield an expected consequence. However, comparative risk typically calls for a broader view. A variety of disciplines contribute insights to risk analysis, as we appreciate by examining Renn's [1] classification of risk perspectives, which follows.

Four perspectives contribute to risk assessment. The actuarial perspective extrapolates from historical circumstances to assign expected human health outcomes based on averages for population cohorts. The epidemiological perspective correlates apparent outcomes with population characteristics and circumstances. The toxicological approach develops explicit causal models of exposure and dose-and-response to characterize expected value outcomes. The engineering approach develops even more explicit causal models to simulate probabilistic outcomes. Environmental risk assessment relies most heavily on the human toxicological approach, although a distinct new ecological risk assessment field has also developed.

Two perspectives contribute to risk management. The economic perspective encourages us to measure expected utility instead of expected consequence, and to weigh both risks and benefits. The psychological perspective highlights the multi-dimensionality of risk and emphasizes that we apply perceptual filters that lead us to behave differently than simple models of rationality would suggest. These perspectives are also valuable when assessing the socioeconomic impacts of environmental threats.

Finally, two perspectives contribute to risk communication. A social perspective reinforces the multi-dimensional view of risk, highlighting fairness issues, and it also makes us aware of social amplification of risk in the mass media. A cultural perspective helps us realize that different people have different framing assumptions, or

basic mental models of the world, and of risk. These perspectives also come into play when considering socioeconomic and quality-of-life impacts of environmental threats.

Comparative risk analysis can be considered as a type of policy analysis, in that it supports tradeoff decisions with broad implications. Good scientific policy analysis arguably should be scientifically adequate, add substantial value, have a visible impact, and be viewed as legitimate by affected parties [2]. Hence, the practice of comparative risk involves much more than technical assessment, and its practitioners need to be familiar with the range of relevant fields.

3. Goals and Uses of CRA

CRA is a flexible process and at a complicated macro scale study, the flexibility may lead to an unfocused, inefficient process. Thus, defining the goal of a CRA is important because the goal shapes the CRA process and its utility. A centralized goal for CRA does not exist although at the state wide and nationwide scales, prioritizing risks has been a major theme. Yet, ideally, the goal of a CRA should go beyond prioritizing risks. In its broadest terms, Jones [3] views "CRA as a dialogue between those who have information about the environment and those who make decisions". Along the same lines Ijjasz and Tlaiye [4] note that other than prioritizing risks and strategies, another goal of CRA is to "promote a structured, fair, and open exchange of ideas among scientists, citizens, and government officials on a broad range of environmental risks using the best available data". Keane and Cho [5] note that being a policy tool, the CRA approach is most "useful for environmental planning and decision making when it is explicitly coupled with risk management and local capacity building". CRA also integrates environmental justice by getting stakeholder input and improves social trust because if the stakeholders are informed of the risks, then they don't need to depend on experts' judgments and can confidently make their own decisions.

According to Hammitt [6] and Davies [7], other uses of CRA include:

- informing policy decisions with better science (building a political consensus around a vision of environmental protection),
- catalyzing and mobilizing opinion about relative risk so that action can be taken,
- educating government officials, stakeholders, public opinion leaders, risk analysts, and citizens,
- being the starting point for setting budgetary and other priorities for environmental agencies,
- supporting the sift of decisions to state and local environmental agencies,
- building trust among different stakeholders,
- focusing people on the questions of what are the benefits of a program or action, and what are we getting for the resources expended,
- identifying neglected problems (e.g. indoor radon),
- making the assumptions behind decisions more transparent (e.g. why not take more action on cigarette smoking), and
- helping identify needed data.

At the macro scale, CRAs have triggered prevention of program cuts (Vermont), directing new resources to new programs (Washington state), focusing attention on a few problems (Northeast Ohio), justification for budgets (Seattle), and elevation of administration initiatives on the policy agenda [8].

In micro applications, CRAs often have more focused objectives within the general goal of evaluating and comparing possible alternatives and their risks in solving problems.

4. The CRA process

A macro scale CRA project generally has two stages: risk comparison and ranking, and strategic analysis and priority setting. Both of these steps require multiple stages where a critical decision has to be made in accordance with the risk management objectives. As much as possible, these decisions on risk should be logically consistent, administratively compatible, equitable, and compatible with cognitive constraints and biases [9].

Some of the decisions related to ranking are the category, definitions, and level of aggregation. For example, depending on the goals of the CRA, the choice of category to be ranked can be (i) the physical or chemical agent responsible (e.g. pesticides, particulates, indoor radon), or (ii) the human activity creating the risk (e.g. coal fired power plants, transportation, pesticide application), or (iii) the exposure pathway (e.g. air, drinking water, food), or (iv) the effects on human health and ecology (cancer, neurotoxic effects to children, risks to fish resources) [6,9].

Another important choice is the level at which we choose to aggregate threats when organizing the analysis. A high level of aggregation reduces the number of threats we must compare, but a low level of aggregation ensures that threats within a category are homogeneous. Inconsistent aggregation leads to arbitrary comparisons. For example air pollution might rank as high but if the class were disaggregated into tropospheric ozone, particulates, carbon monoxide, and other hazardous pollutants, some of these would rank high and others low [6].

In no sense is CRA a purely objective process. Risk comparisons involve a range of assumptions and value judgments, and they rely on factual information of varying incertitude.

Another important step in CRA is what happens after the ranking. Use of CRA results in risk management is not straightforward: "the decision is affected by current management framework, multi-faceted nature of environmental problems, and the potential effectiveness of strategies" [8]. The decision is also affected by "economics, public input, potential for pollution prevention, need to address the existence of disparate impacts on different populations, and emergence of future risks" [3]. One way to improve the decision is to not only rank the baseline risk but also to rank the risk reduction options [10]. This approach may be a way to incorporate cost into the decisions. "Cost benefit analysis is important for decision making and unless risk reduction options are given, the CRA only gives the benefit side of a cost-benefit analysis. One can only do a cost-benefit analysis of a solution not a problem" [7]. But if we do risk reduction options these can be connected. If we move in this direction we

will be faced with the question of willingness to pay. Behavioral patterns for willingness to pay are already emerging [11] and should continue to emerge to make headway in this direction.

Caution should be exercised in focusing on risk comparisons which is often easier than comparing risk reduction opportunities (Adam Finkel cited in [8]). For example, "in focusing attention on 'problem A' versus 'problem B', attention is diverted from better alternative solutions, which might address multiple problems simultaneously".

5. Micro studies

Micro applications of CRA are based on analysis and evaluation of a relatively focused environmental problem. The goal of CRA in micro applications is to compare the risks of alternative solutions to a particular problem. This approach to problem solving and decision making allows consideration of all possible options and preferably incorporation of stakeholder input into the decision making process.

As an example, micro CRA can be used to determine and analyze the options for oil spill response and preparedness. Possible solutions to an oil spill may be natural recovery, on-water recovery, shoreline cleanup, oil and dispersant and on-water in-situ burning. To conduct a CRA, beneficial and undesirable effects of each option are evaluated and ranked. Environmental risks for an oil spill may include ecological damage to terrestrial, shoreline, and subtidal benthic habitats, and water column resources which would have to be analyzed within the scope of their appropriate spatial and temporal scales.

Other micro studies could include comparisons of risks of different chemicals used for a particular purpose, options for management of dredged contaminated sediments, options for managing the west Nile virus, or determining between production of intact and non-intact beef.

6. Macro applications

U.S. government agencies at various levels have logged significant experience with policy-oriented, macro-level CRA. Gutenson [12] suggests that the starting point was a series of Integrated Environmental Management Projects performed during the 1980s. These took place in Santa Clara, CA, Philadelphia, PA, Baltimore, MD, Denver, CO, and the Kanwha Valley, WV. Their common goal was to improve local environmental decision-making by supporting it with quantitative risk analysis. However, it was the national-level Unfinished Business report, described below, that really started the ball rolling.

6.1. HISTORICAL DEVELOPMENT OF PROGRAMMATIC CRA

In 1986, Lee Thomas, then Administrator of the U.S. Environmental Protection Agency (USEPA), asked his management team for a cross-cutting analysis of environmental

threats. He wanted to understand whether his fragmented agency was devoting more resources to the worst threats, as an efficiently designed and managed agency would do. He also wanted his team to scan the horizon to determine what environmental problems were newly emerging or simply remained unaddressed by current policies. The response to this request was the first national comparative risk report, *Unfinished Business*, published in 1987 [13].

The effort involved eighty-two USEPA staff members and a consultant, organized into a management team and four expert work groups. Health scientists staffed the cancer and non-cancer health effects work groups. Biologists and ecologists staffed the ecological effects work group, and economists staffed a welfare effects work group. They systematically evaluated thirty-one environmental threats such as indoor radon, stratospheric ozone depleters, and pesticides, chosen to represent the existing program areas at the USEPA.

Upon completing the analyses, the USEPA team reported several important results. First, the experts' risk ranking differed from the public's perceptions of relative risk as measured in opinion surveys, and from the U.S. Congress' priorities as implied by budget allocations. Second, few risks had high impacts across all endpoints (cancer, non-cancer, ecological, and welfare), instead, different problems affected different endpoints. Third, there was rampant uncertainty forcing the analysts involved to exercise a great deal of expert judgment, making it clear that this exercise involved scientific policy analysis, not rigorous analytical research. For further details, see [14, 15].

6.2 STATE AND LOCAL COMPARATIVE RISK PROJECTS

Unfinished Business created a splash in policy circles, and its success inspired the USEPA to offer grants to encourage U.S. regions, states, and localities to undertake similar projects. From 1988 to 1998, some twenty-four states and more than a dozen localities undertook comparative risk projects. See [14, 15, 16] for more information.

The first few projects (Washington, Vermont, Colorado) received substantial funds from USEPA—several hundred thousand dollars each—to acknowledge their pilot status. These projects combined the lessons learned from the IEMPs with the basic methodology for assessing and comparing risks from *Unfinished Business*: that is, using existing data for the purposes of guiding environmental priority-setting decisions rather than for regulatory standard setting purposes [12]. Subsequent projects received much less federal support, with later projects such as Minnesota and New Jersey receiving grants in the \$50k-\$100k range. All of these projects also depended on substantial state/local funding, plus vast amounts of in-kind resources, especially expert labor.

Table 1 summarizes key characteristics of some of the comparative risk projects that have been performed since 1987. In each of the tables, the projects shown are those for which we were able to find information. Some are old enough that their reports and key personnel could not be tracked down. As can be seen, there are similarities and differences in scope across projects. Similarities include the following:

- Almost every project evaluates human health and ecological impacts or environmental threats.

- Every project evaluates current residual risk, that is, the risk that still remains given decades of environmental regulation and management. A few projects also report trends or provide longer-term outlooks.
- Every project defines its geographic scope contiguously with the boundaries of its sponsoring jurisdiction, although a couple of projects consider extra-territorial impacts attributable to local actions (e.g., climate change due to local fossil fuel consumption).

Differences in scope among projects include the following:

- Although most projects evaluate a third category of impacts, it is variously called "welfare," "quality of life," "economic," and "socioeconomic." The difference is more than semantic, because different impacts are measured in each project.
- The number of threats assessed varies widely. Early projects borrow the USEPA's threat list, but several more recent projects evaluate a larger number of threats, ranging from specific types of toxic emissions to broad changes in land use to economic innovations like factory farming, disaggregating and reaggregating them in several ways over the course of the analysis and rollout efforts.

Table 2 shows additional characteristics of some comparative risk projects, focusing on points where assessment and communication overlap. Only one area of substantial similarity exists among the projects: Every project has a normative focus on efficiency as the key decision criterion. This is to be expected because the risk analysis paradigm targets efficiency. A few projects have a secondary focus on equity or fairness issues.

Dissimilarities shown in Table 2 include the following:

- The projects vary widely in their handling of uncertainty. Most briefly mention their projects' inadequate knowledge base and imprecise risk estimates, but some attempt explicitly to rate the confidence they have in their findings. A very few go further to explore formally the implications of uncertainty on their policy lessons.
- Projects also differ in whether they stop after ranking risks, or whether they take the next step and propose detailed risk management strategies.
- Risk communication strategies also vary widely. Some projects view communication with external stakeholders and the general public as something that only happens at the project's end. Others have heavy public involvement at the very beginning, to help scope the project. A few, mostly the more recent projects, also incorporate the public at intermediate steps, to help with analytical and judgmental tasks.

Projects have tried a variety of recipes for legitimacy [14], as they seek to be seen by the public as desirable, proper, and appropriate [17]. No project wants to be viewed as captured, or incompetent, or unrepresentative, or not in the public interest. Scientific adequacy can contribute to legitimacy. So can an appropriately transparent process with the right participants. Involvement of public officials may be key, because they are constitutionally elected or appointed, and they live under checks and balances

that limit their ability to stray from the public interest. Involvement of major stakeholders may also be crucial, because in a pluralistic society they represent their own interests most effectively. Direct public involvement may also be necessary to reduce the likelihood that the professionals—experts, officials, and special interests—will join forces against the general public. Table 3 shows that most comparative risk projects have relied heavily on experts. The projects vary widely in their involvement of public officials, stakeholders, and the general public.

7. Stakeholder participation (micro and macro)

Keane and Cho [5] note that the most successful CRAs in developing countries have " carefully tailored the scope and complexity of the approach to match local conditions and data and have actively included a broad range of institutions and stake holders from the very beginning of the process". Grumbly [18] noted that "citizen participation was not only an essential part of the decision making process that enhances credibility and accountability but they found it to be economical and effective". Jones [3] suggests that "the utility of any CRA is more dependent on the inclusiveness of participants than on the technical validity of the ranking scheme". On the other hand, while improving public participation, risk assessors should also continue to be included in the process, and the science on risk management, decision-making, and priority setting improved [19]. The public will more likely get involved if they have a sense of ownership [20]. Indeed there is evidence that more intensive stakeholder processes are more likely to result in higher-quality decisions [21]. Participation also provides a means of addressing worry, which affects perceived risks [22].

Presence of social trust is also crucial. If the public doesn't know much about the benefits and risks, then their decision is based on social trust. On the other hand, for cases where people are knowledgeable, there are no significant correlations between social trust and perceived risks and benefits [23,24]. In those cases the social trust is for institutions that are responsible for regulating the risk.

8. Recommendations for Future Projects

Although the USEPA grant program has run its course, interest in comparative risk as a useful policy analysis tool continues to increase. Its cross-cutting perspective provides a valuable complement to indicators and benchmarking efforts, and issue-specific regulatory programs. Its broadly scoped analysis approach complements the narrower, more detailed risk assessments performed in support of specific regulatory proposals. In private industry, many product and service design choices involve environmental tradeoffs, and comparative risk analysis helps inform those tradeoffs.

The experience to date with comparative risk suggests several lessons that designers of future projects should seriously consider:

- There is no standard set of environmental threats that should be compared, and any project that pursues comprehensiveness will fail. Instead, the context of

each specific project should determine its scope. The tables show what other elements of scope have become standardized.

- There is widespread agreement that multiple types of impacts should be assessed. Human health and ecological impacts represent the minimum set, but others also deserve attention.
- Scientific uncertainties are a significant problem, and given the broad scope of comparative risk analysis, it is likely to remain so. Future projects should work with and learn from the uncertainties they encounter.
- In a public setting, the legitimacy of the project matters. Future projects should draw on multiple sources to ensure legitimacy, especially given that inevitable uncertainty diminishes scientific rigor.

TABLE 1: Characteristics of Selected U.S. Comparative Risk Projects Based on a review of project documents by the first author.

Project Name	Dates Performed	Number of Threats Assessed	Classes of Impacts Assessed	Time Horizon of Analysis	Spatial Extent of Analysis
USEPA	1987, 1990	31	Cancer, non-cancer, EC, welfare	Current residual risk	Nation
Washington	1988–90	23	HH, EC, ECN	Current residual risk, trend	State
Vermont	1988–91				State
Colorado	1988–90				State
Louisiana	1990–91				State
Hawaii	1990–92				State
Maine	1990–96	15	HH, EC, QOL	Current residual risk	State
Michigan	1991–92	24	HH, EC	Current residual risk	State
California	1992–94	24	HH, EC, social welfare	Current residual risk	State
New Hampshire	1993–97	53	HH, EC, ECN		State
Florida	1993–95	12	HH, EC, QOL	Current residual risk	State
Kentucky	1993–95	115	HH, EC, QOL	Current residual risk	State
Tennessee	1993–96				State
Alaska	1993–95				State
Texas	1993–96	25	HH, EC, ECN	Current residual risk	State
Mississippi	1994–97				State
Ohio	1994–96	45	HH, EC, QOL	Current residual risk	State
Nebraska	1994–99				State
North Dakota	1994–97				State
Utah	1994–95				State
District of Columbia	1996	8	HH, EC, welfare, public opinion		District
New York	1996–2001	14 (aggregated from 315)	HH, EC, QOL	Current residual risk	State
Minnesota	1996–98	12	HH, EC, QOL	Current residual risk	State
Iowa	1996–98		HH, EC, QOL		State
Arizona	1993-95	14	HH, EC, QOL		State
New Jersey	1998–2002	75	HH, EC, Socioeconomic	Current residual risk, trend	State

Key: HH = Human Health, EC = Ecological, QOL = Quality of Life, ECN = Economic

TABLE 2: Additional Characteristics of Selected U.S. Comparative Risk Projects
Based on a review of project documents by the first author.

Project Name	Uncertainty Explicitly Addressed?	Includes Risk Management Phase?	Normative Focus	Elements in Communication Strategy
USEPA	Yes	No	Efficiency	Rollout
Washington	No	Yes	Efficiency	Rollout
Vermont			Efficiency	Scoping, rollout
Colorado			Efficiency	
Louisiana			Efficiency	
Hawaii			Efficiency	
Maine	No	Yes	Efficiency	Scoping, analysis, rollout
Michigan	No	No	Efficiency	Rollout
California	Yes	No	Efficiency, Equity	None
New Hampshire	Yes	Yes	Efficiency, Equity	Scoping, rollout
Florida	Yes	Yes	Efficiency, Equity	Rollout
Kentucky	No	No	Efficiency	
Tennessee			Efficiency	
Alaska			Efficiency	
Texas	No?	Yes?	Efficiency	Rollout
Mississippi			Efficiency	
Ohio	No	Yes	Efficiency	Scoping, analysis, rollout
Nebraska			Efficiency	
North Dakota			Efficiency	
Utah			Efficiency	
District of Columbia	No	No	Efficiency	Analysis
New York	No	Yes	Efficiency Equity	Analysis, rollout
Minnesota	Yes	No	Efficiency	Analysis, rollout
Iowa			Efficiency	Scoping, analysis, rollout
Arizona	No	No	Efficiency	Scoping, analysis, rollout
New Jersey	Yes	No	Efficiency Equity	Scoping, analysis, rollout

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TABLE 3: Involvement in Selected U.S. Comparative Risk Projects (based on [4])

Project Name	Experts	Officials	Stakeholders	Public
USEPA	High	High	Low	Low
Washington	High	High	High	High
Vermont	Medium	Low	Low	High
Colorado	High	Medium	Medium	Medium
Louisiana	Medium	Medium	High	Medium
Hawaii	High	Medium	Medium	Medium
Maine	High	Medium	High	High
Michigan	High	Low	Medium	Low
California	High	Medium	High	Low
New Hampshire	High	Medium	High	Medium
Florida	High	High	Low	Low
Kentucky	High	High	High	Medium
Tennessee	High	Low	High	Low
Alaska	High	Low	Medium	High
Texas	High	Medium	High	Low
Mississippi	High	High	High	Low
Ohio	Medium	Medium	Medium	High
Nebraska	High	High	Low	Low
North Dakota	Medium	High	High	Medium
Utah	Medium	High	High	Low
District of Columbia	High	Low	Medium	Low
New York	High	Medium	High	Low
Minnesota	Medium	High	Medium	Medium
Iowa	High	Medium	High	Medium
Arizona	High	Medium	High	Medium
New Jersey	High	High	Medium	Low

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MULTI-CRITERIA DECISION ANALYSIS: A FRAMEWORK FOR STRUCTURING REMEDIAL DECISIONS AT CONTAMINATED SITES

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Abstract

Decision-making in environmental projects is typically a complex and confusing exercise, characterized by trade-offs between socio-political, environmental, and economic impacts. Cost-benefit analyses are often used, occasionally in concert with comparative risk assessment, to choose between competing project alternatives. The selection of appropriate remedial and abatement policies for contaminated sites, land-use planning and other regulatory decision-making problems for contaminated sites involves multiple criteria such as cost, benefit, environmental impact, safety, and risk. Some of these criteria cannot easily be condensed into a monetary value, which complicates the integration problem inherent to making comparisons and trade-offs. Even if it were possible to convert criteria rankings into a common unit this approach would not always be desirable since stakeholder preferences may be lost in the process. Furthermore, environmental concerns often involve ethical and moral principles that may not be related to any economic use or value.

Considerable research in the area of multi criteria decision analysis (MCDA) has made available practical methods for applying scientific decision theoretical approaches to multi-criteria problems. However, these methods have not been formalized into a framework readily applicable to environmental projects dealing with contaminated and disturbed sites where risk assessment and stakeholder participation are of crucial concern. This paper presents a review of available literature on the application of MCDA in environmental projects. Based on this review, the paper develops a decision analytic framework specifically tailored to deal with decision making at contaminated sites.

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1. Current and Evolving Decision-Analysis Methodologies

Environmental decisions are often complex, multi-faceted, and involve many different stakeholders with different priorities or objectives – presenting exactly the type of problem that behavioral decision research shows humans are typically quite bad at solving, unaided. Most people, when confronted with such a problem will attempt to use intuitive or heuristic approaches to simplify complexity until the problem seems more manageable. In the process, important information may be lost, opposing points of view may be discarded, elements of uncertainty may be ignored -- in short, there are many reasons to expect that, on their own, individuals (either lay or expert) will often experience difficulty making informed, thoughtful choices about complex issues involving uncertainties and value tradeoffs (McDaniels et al., 1999).

Moreover, environmental decisions typically draw upon *multidisciplinary* knowledge bases, incorporating natural, physical, and social sciences, medicine, politics, and ethics. This fact, and the tendency of environmental issues to involve shared resources and broad constituencies, means that *group* decision processes are called for. These may have some advantages over individual processes: more perspectives may be put forward for consideration, the chances of having natural systematic thinkers involved is higher, and groups may be able to rely upon the more deliberative, well-informed members. However, groups are also susceptible to the tendency to establish entrenched positions (defeating compromise initiatives) or to prematurely adopt a common perspective that excludes contrary information -- a tendency termed "group think." (McDaniels et al., 1999).

For environmental management projects, decision makers may currently receive four types of technical input: modeling/monitoring, risk analysis, cost or cost-benefit analysis, and stakeholders' preferences (Figure 1a). However, current decision processes typically offer little guidance on how to integrate or judge the relative importance of information from each source. Also, information comes in different forms. While modeling and monitoring results are usually presented as quantitative estimates, risk assessment and cost-benefit analyses incorporate a higher degree of qualitative judgment by the project team. Only recently have environmental modeling (such as fate and transport models) and formalized risk assessment been coupled to present partially integrated analyses to the decision-maker (e.g., Army Risk Assessment Modeling project, ARAMS (Dortch, 2000). Structured information about stakeholder preferences may not be presented to the decision-maker at all, and may be handled in an *ad hoc* or subjective manner that exacerbates the difficulty of defending the decision process as reliable and fair. Moreover, where structured approaches *are* employed, they may be perceived as lacking the flexibility to adapt to localized concerns or faithfully represent minority viewpoints. A systematic methodology to combine these inputs with cost/benefit information and stakeholder views to rank project alternatives has not yet been developed. As a result, the decision maker may not be able to utilize all available and necessary information in choosing between identified remedial and abatement alternatives.

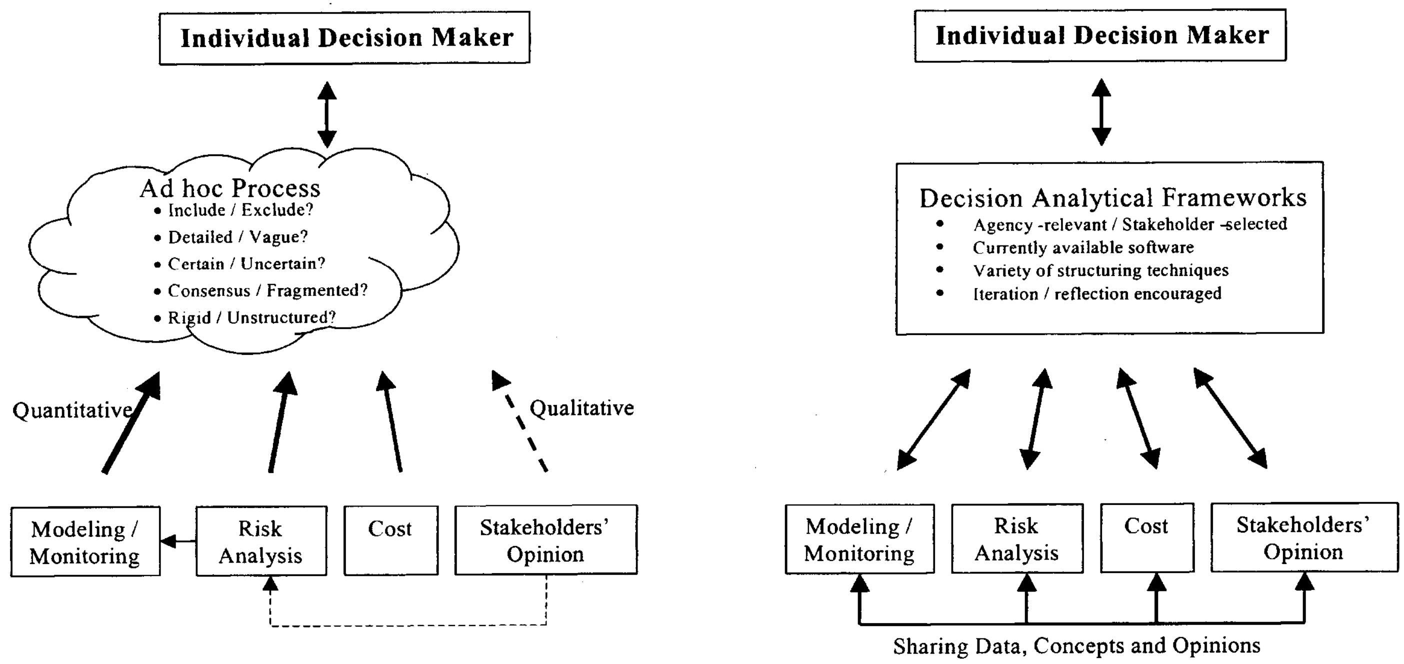
In response to current decision-making challenges, this paper develops a systematic framework for synthesizing quantitative and qualitative information that builds on the recent efforts of several government agencies and individual scientists to

implement new concepts in decision analysis and operations research. This will help to both facilitate analysis and provide for more robust treatment of stakeholder concerns. The general trends in the field are reflected in Figure 1b. Decision analytical frameworks may be tailored to the needs of the individual decision maker or relate to multiple stakeholders. For individual decision-makers, risk-based decision analysis quantifies value judgments, scores different project alternatives on the criteria of interest, and facilitates selection of a preferred course of action. For group problems, the process of quantifying stakeholder preferences may be more intensive, often incorporating aspects of group decision-making. One of the advantages of an MCDA approach in group decisions is the capacity for calling attention to similarities or potential areas of conflict between stakeholders with different views, which results in a more complete understanding of the values held by others. In developing this framework, the paper will draw from existing literature on environmental applications of multi criteria decision theory and regulatory guidance developed by the US and international agencies.

2. MCDA Methods and Tools

MCDA methods evolved as a response to the observed inability of people to effectively analyze multiple streams of dissimilar information. There are *many* different MCDA methods. They are based on different theoretical foundations such as optimization, goal aspiration, or outranking, or a combination of these:

- **Optimization models** employ numerical scores to communicate the merit of one option in comparison to others on a single scale. Scores are developed from the performance of alternatives with respect to an individual criterion and then aggregated into an overall score. Individual scores may be simply added up or averaged, or a weighting mechanism can be used to favor some criteria more heavily than others. Typically, (but not always, depending upon the sophistication of the objective function) good performance on some criteria can compensate for poor performance on others. Normalizing to an appropriate single scale may be problematic. Consequently, optimization models are best applied when objectives are narrow, clearly defined, and easily measured and aggregated. Considerable research and methods development has been done on multiobjective optimization. This work has mostly involved finding the "Pareto frontier", along which no further improvements can be made in any of the objectives without making at least one of the other objectives worse (Diwekar and Small, 2002).
- **Goal aspiration,** reference level, or threshold models rely on establishing desirable or satisfactory levels of achievement for each criterion. These processes seek to discover options that are closest to achieving, but not always surpassing, these goals. When it is impossible to achieve all stated goals, a goal model can be cast in the form of an optimization problem in which the decision maker attempts to minimize the shortfalls, ignoring exceedances. To this extent, overperformance on one criterion may not compensate for underperformance on



a. Current Decision-Making Process

b. Evolving Decision-Making Process

Figure 1: Current and evolving decision -making processes for contaminated sediment management.

others. Alternatively, the decision maker may seek to satisfy as many of the goals as possible (even if only just barely) and ignore the fact that some performance metrics may be *very* far from target levels. Goal models are most useful when all the relevant goals of a project cannot be met at once.

- **Outranking models** compare the performance of two (or more) alternatives at a time, initially in terms of each criterion, to identify the extent to which a preference for one over the other can be asserted. In aggregating preference information across all relevant criteria, the outranking model seeks to establish the strength of evidence favoring selection of one alternative over another -- for example by favoring the alternative that performs the best on the greatest number of criteria. Outranking models are appropriate when criteria metrics are *not* easily aggregated, measurement scales vary over wide ranges, and units are incommensurate or incomparable. Like most MCDA methods, outranking models are *partially* compensatory (Guitouni and Martel 1998).

The common purpose of these diverse methods is to be able to evaluate and choose among alternatives based on multiple criteria using systematic analysis that overcomes the observed limitations of unstructured individual and group decision-making. Different methods require different types of raw data and follow different optimization algorithms. Some techniques rank options, some identify a single optimal alternative, some provide an incomplete ranking, and others differentiate between acceptable and unacceptable alternatives.

An overview of four principal MCDA approaches is provided in the remainder of this section. A more detailed analysis of the theoretical foundations of these methods and their comparative strengths and weaknesses can be found in Belton and Stewart (2002) and other references.

2.1 ELEMENTARY METHODS

Elementary methods are intended to reduce complex problems to a singular basis for selection of a preferred alternative. Competing decision criteria may be present, but intercriteria weightings are not required. For example, an elementary goal aspiration approach may rank alternatives in relation to the total number of performance thresholds met or exceeded. While elementary approaches are simple and analysis can, in most cases, be executed without the help of computer software, these methods are best suited for single-decision maker problems with few alternatives and criteria – a condition that is rarely characteristic of environmental challenges.

2.1.1. Pros and Cons Analysis A Pros and Cons Analysis is a qualitative comparison method in which experts identify the qualities and defects of each alternative. The lists of pros and cons are compared to one another for each alternative, and the alternative with the strongest pros and weakest cons is selected. Pros and Cons Analysis is suitable for simple decisions with few alternatives (2 to 4) and few discriminating criteria (1 to 5) of approximately equal value. It can be implemented rapidly. (DOE, 2001) Other methods are based on the Pros and Cons concept, including SWOT Analysis and Force Field Analysis. SWOT stands for Strengths, Weaknesses, Opportunities, and Threats. SWOT analysis helps reveal changes that can be usefully made. In Force Field Analysis

the viability of a project is evaluated by comparing the forces for and against the project.

2.1.2. Maximin and Maximax Methods The maximin method is based upon a strategy that seeks to avoid the worst possible performance – or "maximizing" the poorest ("minimal") performing criterion. This is achieved by assigning total importance to the criteria in which an alternative performs the worst, ranking all alternatives by the strength of their *weakest* attribute. The alternative for which the score of its weakest attribute is the highest is preferred. In multi-attribute decision-making the maximin method can be used only when all attributes are comparable so that they can be measured on a common scale, which may present a serious limitation. An analogous strategy called *maximax* ranks alternatives solely by their *best* performing criterion. Maximin and minimax are noncompensatory, in that individual alternative performance is judged on the basis of a single criterion (although different criteria may be selected for different alternatives). Minimax and minimin methods also exist. Their names make their underlying concepts self-explanatory.

2.1.3. Conjunctive and Disjunctive Methods The conjunctive and disjunctive methods are non-compensatory, goal aspiration screening methods. They do not require attributes to be measured in commensurate units. These methods require satisfactory (in comparison with a predefined threshold) rather than best possible performance in each attribute -- i.e. if an alternative passes the screening, it's acceptable. The underlying principle of the conjunctive method is that an alternative must meet a minimum cutoff level (called a performance threshold) for *all* attributes. The disjunctive method is a complementary method. It requires that an alternative should exceed the given thresholds for at least *one* attribute. These simple screening rules can be used to select a subset of alternatives for analysis by other, more complex decision-making tools, or provide a basis for selection in and of themselves as in a strategy called Elimination by Aspects. In this approach, performance criteria are ordered in terms of importance. Alternatives that fail to meet the most important threshold level are discarded. Remaining alternatives are then tested against the second most important criteria, and on down. The last alternative to be discarded (in the event no alternative meets *all* criteria) is preferred.

2.1.4. Lexicographic Method A lexicographic analysis of any problem involves a sequential elimination process that is continued until either a unique solution is found or all the problems are solved. In the lexicographic decision-making method attributes are first rank-ordered in terms of importance. The alternative with the best performance on the most important attribute is chosen. If there are ties with respect to this attribute (which is quite likely if many alternatives are considered), the performance of the tied alternatives on the next most important attribute will be compared, and so on, till a unique alternative is found.

It should be noted that in multi-attribute decision-making problems with few alternatives, quantitative input data, and negligible uncertainty, the lexicographic method ends up becoming a selection method based on a single attribute.

2.1.5. Decision Tree Analysis Decision trees are useful tools for making decisions where a lot of complex quantitative information needs to be taken into account (e.g. deciding whether to take immediate action or to postpone action in treating a contaminated groundwater problem -- Wang et al., 2002). The principle behind decision tree analysis is link specific outcomes (or consequences) to specific decision nodes. Decision trees provide an effective structure in which alternative decisions and the consequences of those decisions can be laid down and evaluated. They also help in forming an accurate, balanced picture of the risks and rewards that can result from a particular choice, especially when outcomes may be dependent upon independent choices made by more than one decision maker (as in game theory problems). A basic limitation of the decision tree representation is that only relatively simple models can be shown at the required level of detail since every additional criterion expands the tree exponentially.

2.1.6. Influence Diagrams An Influence Diagram is a graphic representation of a decision problem. This representation provides a framework for building decision-analysis problems but does not provide a framework for quantitative evaluation, unlike Decision Tree Analysis. The graphical representation comprises nodes that represent criteria relevant to the problem. The arrows connecting the nodes represent information flows. The layout of an influence diagram allows a clear representation of dependencies between various nodes. Influence diagrams have been employed to highlight key differences between how environmental problems are perceived by different groups – such as expert and lay stakeholders (Morgan et al., 2002).

2.2 MULTI-ATTRIBUTE UTILITY/VALUE THEORY (MAUT/MAVT)

Multi-Attribute Utility Theory (MAUT/MAVT) is a technique for formally drawing multiple perspectives and evaluations into a decision-making process. The goal of MAUT/MAVT is to find a simple expression for the decision-maker's preferences. Through the use of utility/value functions, this method transforms diverse criteria (such as costs, risks, benefits, stakeholder values) into one common dimensionless scale (utility/value). It also relies on the assumptions that the decision-maker is rational: more is preferred to less, preferences do no change, the decision-maker has perfect knowledge, and the preferences are transitive. The goal of decision-makers in this process is to maximize utility/value, which is a compensatory optimization approach.

The first step in MAUT/MAVT analysis is development of an attribute tree that summarizes the key values to be taken into account. The attribute tree splits top-level objectives into finer attributes and criteria. In the case of applied problems such as management of contaminated sites, criteria at the lowest level should be measurable. The second step in a MAUT/MAVT application is defining criteria and associated weights. The next question concerns the form of the multi-attribute utility function that adjusts the difference between outcomes so that decision-maker's risk attitude is also encoded. The utility graphs could be created based on the data for each criterion. Alternatively, a simple MAU/MAV function may be constructed using a functional dependence (e.g., French et al., 1998, used an exponential function for nuclear plan risks).

The two main considerations used in MAUT/MAVT are: how *great* is an effect (score) and how *important* (weight) is the criterion measured, relative to all other criteria. MAUT/MAVT describes a system of assigning scores to individual effects (e.g. human health risk reduction), which are then combined into overall aggregates on the basis of a perceived weighting of each score. It brings together different considerations in a structured way. Differences in the importance of goals are attributed to the particular interests of the affected parties and the decision makers. The technique makes these differences, and similarities, lucid by eliciting from participants their subjective judgments about the importance of outcomes and using these as a basis for comparison. Thus, by taking the decision-maker's preferences into consideration, criteria can be weighted by importance. MAUT/MAVT leads to a complete ranking of all the alternatives based on the decision-maker's preferences. The goal is not to reach a forced "consensus" through averaging different stakeholder weightings, but to clarify positions and to test the feasibility of various policy objectives.

Concerns for the practical implementability of MAUT/MAVT led to the development of the Simple Multi Attribute Rating Technique (SMART). SMART is a simplified multi-attribute rating approach which utilizes *simple* utility relationships. Data normalization to define the MAUT/MAVT/SMART utility functions can be performed using any convenient scale. The SMART methodology allows for use of less of the scale range if the data do not discriminate adequately so that, for example, alternatives, which are not significantly different for a particular criterion, can be scored equally. Research has demonstrated that simplified MAUT/MAVT decision analysis methods are robust and replicate decisions made from more complex MAUT/MAVT analysis with a high degree of confidence (Baker et al., 2001).

2.3. ANALYTICAL HIERARCHY PROCESS

The Analytical Hierarchy Process (AHP), developed by Thomas Saaty in 1980, is a quantitative comparison method used to select the optimal alternative by comparing project *alternatives* (e.g. methods for disposing of contaminated sediments) based on their relative performance on the *criteria* of interest (e.g. impact on ecological habitat, project costs, etc.), after accounting for the decision-maker's relative *preference* or *weighting* of these criteria. Similar to MAUT, AHP completely aggregates various facets of the decision problem into a single objective function. The goal is to select the alternative that results in the greatest value of the objective function. Like MAUT, AHP is a compensatory optimization approach. However, AHP uses a quantitative comparison method that is based on pair-wise comparisons of decision criteria, rather than utility and weighting functions. Evaluators' express the intensity of a preference for one criterion versus another using a nine-point scale[1]:

1: if the two elements are **equally** important

3: if one element is **weakly/moderately** more important than the other element

5: if one element is **strongly** more important than the other element

7: if one element is **very strongly** more important than the other element

[1] Other scales have been also suggested.

9: if one element is **absolutely/extremely** more important than the other element

All individual criteria must be paired against all others and the results compiled in matrix form. If criterion A is strongly more important compared to criterion B (i.e. a value of 5), then criterion B has a value of 1/5 compared to criterion A. Thus for each comparative score provided, the reciprocal score is awarded to the opposite relationship. The 'priority vector' (i.e. the normalized weight) is calculated for each criterion using the geometric mean of each row in the matrix divided by the sum of the geometric means of all the criteria. The AHP technique thus relies on the supposition that humans are more capable of making relative judgments than absolute judgments. The rationality assumption in AHP is more relaxed than in MAUT. For example, some level of intransitivity (in which A is preferred 5 times than B and B 5 times than C, but C is not assessed at 1/25 A) can be tolerated.

2.4. OUTRANKING

Unlike MAUT and AHP, outranking is based on the principle that one alternative may have a degree of *dominance* over another (Kangas, 2001), rather than the supposition that a single best alternative can be identified. The concept was defined by B. Roy in the 1970s as follows:

The performance of alternatives on each criterion is compared in pairs. Alternative *a* is said to *outrank* alternative *b* if it performs better on some criteria (or, if the criteria are weighted, performs better on the significant criteria) and at least as well as *b* on all others. An alternative that is inferior in some respects and no better than equal in others, is said to be *dominated*; that is, no combination of weightings would suggest a preference for a dominated alternative. Conversely, a *dominant* alternative is superior or equal in all respects. Based on Roy's fundamental partial comparability axiom, all outranking approaches permit incomparability and intransitivity of preferences (unlike in MAUT); however, methods may differ in the way they formalize mathematical approaches (Gal et al., 1999).

In general, outranking investigates relations between alternatives given the preferences of the decision maker and performance of the alternatives on each criterion. (Vincke, 1992). For alternative pairs a and *b*, preferences are expressed for each criterion as one of the following types (Schreck, 2002):

- *a*P*b*, strict preference *a* over *b*.
- *a*Q*b*, weak preference for *a* over *b*.
- *a*I*b*, indifference between the two actions.
- *a*J*b*, inability or refusal to compare the actions.

Preference and indifference thresholds are introduced for each criterion to avoid exaggerating the importance of small differences in performance. The indifference threshold is the difference beneath which a decision-maker has no preference – that is, a difference that is too small to be used as a basis of distinction between the two. The preference threshold is the difference above which the decision-maker strongly prefers one management alternative to another. Put in a different way, a preference threshold is the smallest value that is decisive when comparing two actions,

while the highest value of no preference represents the indifference threshold. Between the indifference and preference threshold, weak (or fuzzy) preferences may be represented by any number of mathematical interpolation functions such as linear, step-wise, or Gaussian. The combination of the preference threshold, interpolation function, and indifference threshold is called the *preference function* that describes inter-criterion relations.

Preference functions can take different forms for different criteria or stakeholders, but certain forms may be suggested by the metrics used to assess performance in any of the criteria. For example, quantitative criteria (such as cost) may result in linear or Gaussian preference functions, where as semiquantitative (such as High, Middle, Low scales) may result in step-wise functions. One of the strengths of an outranking approach compared to MAUT and AHP is the ease with which semi- or non-quantitative (e.g., "Would you prefer blue or green?") information can be handled.

Outranking is a partially compensatory method that does not rely upon optimization. The emphasis is on understanding trade-offs and facilitating a structured, quantitative comparison of strengths and weaknesses. Outranking methods allow for intransitivities in criteria weightings and for alternatives that are not considered comparable. For example, *a* may be superior to all other alternatives in several respects, but inferior to all alternatives in others. Selecting *a* over an alternative *b* that performs well (but not dominantly) in all respects depends upon a strongly held preference for the criteria favorable to *a* — a confidence that not all stakeholders may be willing to express. In this case, the two alternatives are said to be *incomparable*. Consequently, the ordering of alternatives provided by outranking methods may be incomplete.

As with other MCDA methods, multiple points of view can be represented in outranking approaches by representing different stakeholders with different intercriteria weightings. The sensitivity of alternative ordering to intercriteria weightings is generally simple to investigate by calculating a *stability interval* over which any one weighting may be adjusted without altering the ordering outcome. Also, outranking methods allow stakeholders to "change their minds" by adjusting intercriteria weightings, or by introducing new criteria or alternatives at any time during the analysis – a flexibility that is much more difficult to introduce into MAUT or AHP.

3. Regulatory Basis for MCDA

This section presents the methodology of decision process implementation in different agencies in the US and Europe. Decision process implementation is often based on physical modeling and engineering optimization schemes. Even though federal agencies are required to consider social and political factors, the typical decision analysis process does not provide specifically for explicit consideration of such issues. Little effort is made to accommodate and understand stakeholder perspectives or to allow for potential learning among stakeholders. On the contrary, the process tends to quickly become adversarial where there is little incentive to understand perspectives and to share information. Nevertheless, our review of regulatory and guidance documents revealed

several programs where agencies are beginning to implement formal decision analytical tools (such as multicriteria decision analysis) in environmental decision making.

3.1. U.S. ARMY CORPS OF ENGINEERS

3.1.1 Role of Decision Analysis in USACE Decision making. Historically, the U.S. Army Corps of Engineers has used essentially a single measure approach to decision-making. The Corps has primarily used net NED[1] Benefits as the single measure to choose among different alternatives. The method makes use of a complex analysis of each alternative to determine the benefits and costs in terms of dollars, and other non-dollar measures (environmental quality, safety, etc.); the alternative with the highest net NED benefit is usually chosen. The *Economic and Environmental Principles and Guidelines for Water and Related Land Resources Implementation Studies* (also known as Principles and Guidelines or P&G) and Engineering Regulation (ER) 105-2-100, *Guidance for Conducting Civil Works Planning Studies* sets out the following six-step planning process:

1. Specification of the water and related land resource problems and opportunities
2. Inventory, forecast and analysis of water and related land resource conditions within the planning area relevant to the identified problems and opportunities.
3. Formulation of alternative plans.
4. Evaluation of the effects of the alternative plans.
5. Comparison of alternative plans.
6. Selection of a recommended plan based upon the comparison of alternative plans.

While the P&G method is not specifically required for planning efforts related to military installation operation and maintenance, regulatory actions or operational/maintenance dredging, it presents a general planning method that influences most USACE decisions. The USACE planning approach is essentially a mono-criterion approach, where a decision is based on a comparison of alternatives using one or two factors (Cost Benefit Analysis is an example of a mono-criterion approach). The P&G approach is deficient in that knowledge of the costs, benefits, impacts, and interactions is rarely precise. This single-number approach is limiting and may not always lead to an alternative satisfactory to key stakeholders. In response to a USACE request for a review of P&G planning procedures, the National Research Council (1999) provided recommendations for streamlining planning processes, revising P&G guidelines, analyzing cost-sharing requirements and estimating the effects of risk and uncertainty integration in the planning process.

Recent implementation guidance of the Environmental Operating Principles (EOP) (http://www.hq.usace.army.mil/cepa/envprinciples.htm) within USACE civil works planning has dictated that projects adhere to a concept of environmental sustainability that is defined as "a synergistic process whereby environmental and economic considerations are effectively balanced through the life of project planning, design, construction, operation and maintenance to improve the quality of life for

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[1] National Economic Development

present and future generations" (USACE 2003a, p. 5). Within some USACE Districts, program management plans for EOP implementation have been drafted (USACE-SAJ, 2002). In addition, revised planning procedures have been proposed to formulate more sustainable options through "combined" economic development/ecosystem restoration plans (USACE, 2003b). While still adhering to the overall P&G methodology, USACE (2003b) advises project delivery teams to formulate acceptable, combined economic development/ecosystem restoration alternatives through the multi-criteria/trade-off methodology shown in Figure 2.

3.1.2. Decision Models Used by USACE. Currently, the USACE uses a variety of mechanistic/deterministic fate and transport models to provide information in quantifying the various economic development/ecological restoration accounting requirements as dictated by P&G-related procedures. The complexity and scope of these models are determined by the various planning teams. Issues such as uncertainty and risk are also addressed through formulation at the individual project management level. As a integration mechanism, the National Research Council (1999) review recommended that further decision analysis tools be implemented to aid in the comparison and quantification of environmental benefits from restoration, flood damage reduction and navigation projects.

In addition to the formulation of various planning alternatives, USACE civil planning and decision processes are being revised to include a system-wide perspective (USACE, 2002). This basin-level focus has its own related decision and planning challenges. Watershed planning and management is complex and cannot be handled by a single state or federal agency. Coordination with agencies at federal, state, and local levels is necessary to assure that all environmental and social obligations are met, and is mandated by virtually all federal natural resource and environmental law (Cole and Feather, 2002). The diverse interests involved in such efforts are rarely totally compatible, and tradeoffs are commonly necessary. Past lumping of environmental objectives under some general rubric, such as environmental protection and ecosystem restoration, has too often generated incompletely satisfying consequences (Cole and Feather, 2002). As a result of the challenges associated with the watershed perspective, the National Research Council (1999) recommended that additional resources be allocated for long term monitoring on larger-scale water resources projects.

3.1.3. MCDA Tools Recommended for Use by USACE. The Corps has recently been restructuring its planning processes to include multi-criteria approaches for planning and decision-making. Additional criteria, such as environmental restoration, are being considered in a multi-criterion approach to decision-making (Males, 2002). In *Trade-Off Analysis Planning and Procedures Guidebook, 2002,* a report written for the U.S. Army Corps of Engineers, Charles Yoe lays out a multi-criterion decision analytic approach for comparing and deciding between alternative plans. The report relates the P&G, six-step planning process described above to a multi-decision analytic approach, as depicted in Figure 2.

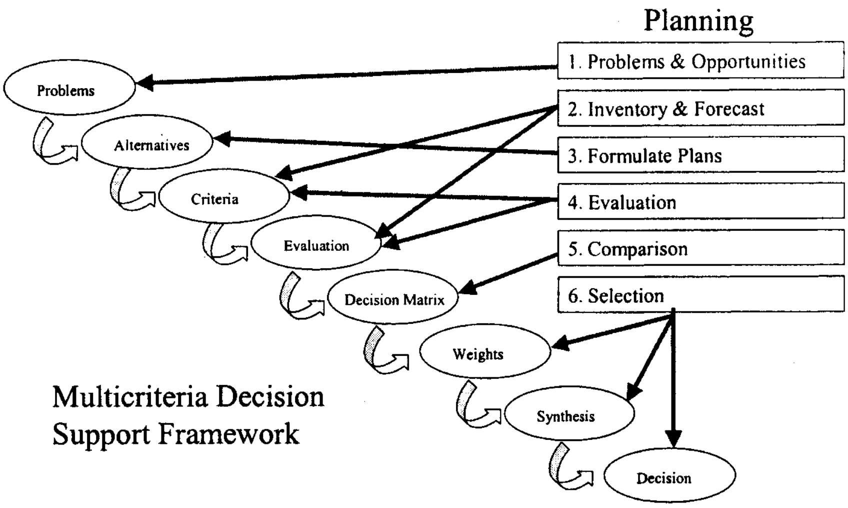

Figure 2: Relation of planning process to multi-criteria decision support framework (Yoe, 2002). The rectangle shows the current planning process practiced in the Corps. The bubbles show steps that are generally a part of standard MCDA techniques.

Other related stakeholder/model/decision tools have been developed by the USACE-Institute of Water Resources (IWR). The Shared Vision Planning (SVP) methodology utilizes stakeholder-based discussions and modeling with the STELLA™ simulation software package. Two primary examples of the use of SVP/Stella include the National Drought Study (Werick and Whipple, 1994) and a basin-scale water management assessment for the Apalachicola/Flint River watersheds (Werick et al., 1996).

Despite the existence of new guidance and policy revisions on the application of multiple criteria decision analytic techniques to environmental projects, there remains a need for a systematic decision analytic framework to implement these methods within the Corps.

3.2. EPA

3.2.1. Role of Decision Analysis in EPA Decision Making. Stahl (2002, 2003) has recently reviewed the decision analysis process in EPA. Stahl observes that although EPA has a mandate to make decisions in the public interest pertaining to the protection of human health and welfare, there are analytical barriers in the EPA process that discourage stakeholder participation, integration of perspectives, learning about new alternatives, and consensus building. Figure 3 illustrates the typical EPA decision analysis process, which is initiated by a legal or regulatory mandate or by stakeholder complaints. According to Stahl:

- The framing process usually conforms to EPA's organizational paradigm, and does not recognize different stakeholder perspectives.

- The problem formulation process is influenced explicitly and implicitly by political factors.
- The process isolates the physical science from social science.
- EPA policy analysts infuse non-quantitative social and political concerns often to rule out options that are considered unacceptable by EPA.

Stahl concludes that this approach compromises the cohesive analysis of human/ecological impacts and frequently results in decisions supportive of the interests of the most powerful stakeholders.

3.2.2. Decision Models Used by EPA. EPA currently uses a variety of modeling tools to support its decision-making process. The majority of these tools are "quantitative multimedia systems that assess benefits/risks associated with each proposed alternative with the objective of selecting the "best option" (Stahl, 2003). EPA models currently are largely deterministic and do not model uncertainty and variability. EPA's economic models typically rely on cost-benefit analysis.

3.2.3. MCDA Tools Recommended for Use by EPA. Our review has identified several guidance documents that introduce decision-analytical tools and recommend their use.

Multi-criteria Integrated Resource Assessment (MIRA) is being proposed as an alternative framework to existing decision analytic approaches at the U.S. EPA (Stahl et al., 2002, Stahl, 2003, USEPA, 2002). MIRA is a process that directs stakeholders to organize scientific data, establish links between the results produced by the research community and applications in the regulatory community. MIRA also encompasses a tool that utilizes AHP-based tradeoff analysis to determine the relative importance of decision criteria. The decision maker's preferences/judgments are obtained first in Expert Choice™ software and then used in the decision analysis spreadsheet to produce decision alternatives. MIRA was developed by EPA Region 3's Air Protection Division as an effort to link its decision to environmental impacts. MIRA is being currently used to develop response to a request for a preliminary analysis of the 8-hour ozone designations. Specifically, MIRA was used to help decision-makers rank counties based on attainment as well as to incorporate scientific data and social values.

Life Cycle Assessment Guidance. Multi-attribute product evaluation is inherent in the nature of life cycle assessment that has been rapidly emerging as a tool to analyze and assess the environmental impacts associated with a product, process, or service (Miettinen & Hamalainen, 1997, Seppala et al 2002). The U.S. Environmental Protection Agency developed the *Framework for Responsible Environmental Decision-Making* (FRED) to assist the Agency's Office of Pollution Prevention and Toxics in their development of guidelines for promoting the use of environmentally preferable products and services (EPA, 2000). The FRED decision-making methodology provides the foundation for linking the life cycle indicator results with technical and economic factors for decision-makers when quantifying the environmental performance of competing products. The FRED life cycle assessment (LCA) approach can be applied to determine and compare the environmental and human health impacts of competing products. The framework identifies data collection needs and describes how to calculate numeric impact indicators for a given product/service across 8 impact categories: global

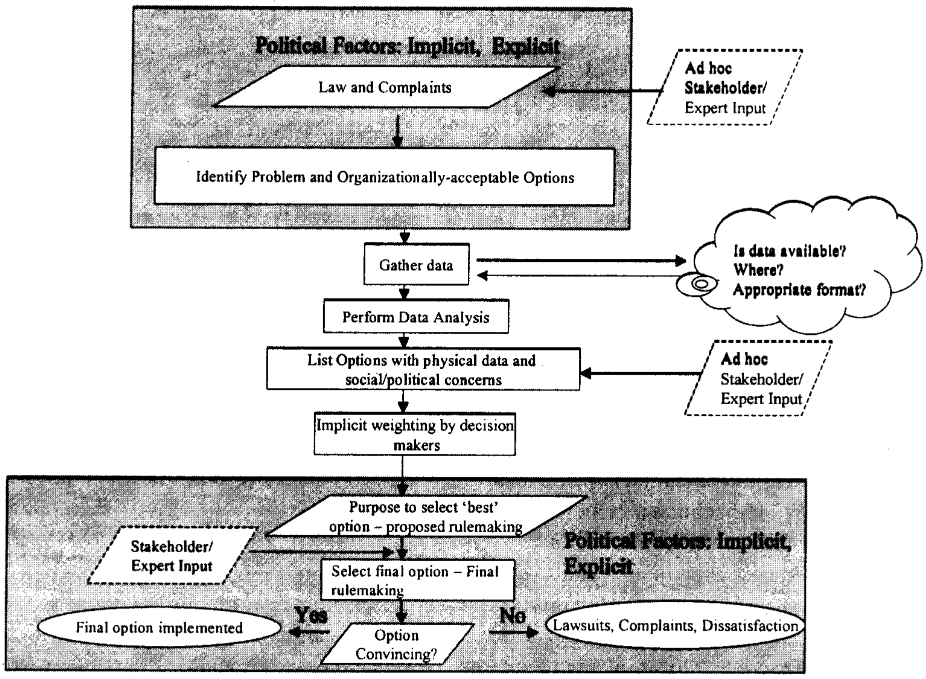

Figure 3. EPA Decision Analysis Process (after Stahl, 2003)

climate change, stratospheric ozone depletion, acidification, photochemical smog formation, eutrophication, human health, ecological health, and resource depletion. The FRED LCA methodology provides the ability for procurement officials and vendors to apply a greater degree of specificity and completeness to the evaluation of competing products or services. It simplifies data collection and impact assessment, making the approach easier and faster to conduct than a detailed LCA. EPA recommends the following approaches to elicit value judgments from stakeholders in order to establish the relative significance (i.e. weights) of the indicators: *Analytical Hierarchy Process, Modified Delphi Technique* and *Multi-Attribute Utility Theory.* The guidance presents case studies for three product categories: motor oil, wall insulation, and asphalt coating.

3.3. DOE

3.3.1 Role of Decision Analysis in DOE Decision Making. Many major decisions for the DOE Environmental Management program (responsible for site clean up) are made in compliance with NEPA and CERCLA regulations and are supported by environmental documentation such as risk assessments and environmental impact assessments. A recent Top-to-Bottom Review of the Department of Energy's cleanup

program (Top-to-Bottom Review Team, 2002) found that the program had fundamental flaws and needed significant change. One of the major findings was that DOE cleanup was not based on comprehensive, coherent, technically supported risk prioritization. Often it was primarily driven by unrealistic concern about litigation rather than by defining the appropriate level of analysis for environmental impacts required to meet program goals and decision-making needs. Inappropriate definition of alternatives could result in reanalysis. The result has been a cleanup approach that has achieved little real risk reduction relative to the funds expended (Top-to-Bottom Review Team, 2002).

In response to this finding, DOE issued a policy that focuses Department officials on conducting cleanup that is aimed at, and achieves, clearly defined, risk-based end states that are representations of site conditions and associated information (DOE, 2003). The implementation plan developed in support of this policy (Corporate Project 7 Team, 2003b) calls for clear documentation of the complex system of values, factors, and activities that were involved in arriving at the remedy decision. This documentation should provide clarity to DOE and stakeholders regarding the rationale for the decision, and it should force all parties to be clear about what they are demanding. The Implementation Plan explicitly calls for analyzing risk trade-offs, or risk balancing for a number of factors:

- ecological and human health;
- worker and public health;
- spatial extent and location of cleanup within a site;
- sequencing of cleanup options for a specific site;
- relative risk to species (plants vs. animals, one animal vs. another);
- resource allocation among alternative cleanup projects; and
- current vs. future risks and short-term versus long-term risks.

The Implementation Plan calls for using risk tradeoffs across the DOE complex (i.e. all DOE establishments) to prioritize among sites. Is it better to clean up smaller sites with little contamination, small sites with greater contamination, or larger sites with greater contamination, and in what order? While risks to human and ecological receptors enter into balancing across sites, environmental and social equity are additional key factors in risk evaluations.

3.3.2. Decision Models Used by DOE. Similar to EPA, DOE uses a variety of multimedia models to support its decision-making process. A Recent review (Corporate Project 7 Team, 2003a) concluded that even though there are a significant number of guidance documents, systems, and processes in use within the DOE complex to determine, manage, and communicate risk, there is a great need for comparative risk assessment tools, risk management decision trees and risk communication tools that would allow site managers to reach agreement with their regulators and other stakeholders, while achieving mutual understanding of the relationship between risk parameters, regulatory constraints and cleanup. Because of DOE mandates, many DOE models are developed specifically for dealing with radiologically contaminated sites and sites with dual (chemical and radiological) contamination. Many models are

deterministic, although probabilistic multimedia models are also used (RESRAD, 2003).

3.3.3. MCDA Tools Recommended for Use by DOE. Our review has identified several guidance documents that introduce decision-analytical tools and recommend their use.

Guide-book to Decision-Making Methods (Baker et al., 2001). This generic guidance developed for a wide variety of DOE decision needs, breaks the decision process into 8 sequential steps: defining the problem, determining the requirements, establishing the goals of the project, identifying alternative methods/products, defining the criteria of concern, selecting an appropriate decision-making tool for the particular situation, evaluating the alternatives against the criteria, and finally validating solution(s) against the problem statement (Figure 4). The report then focuses on how to select a decision-making tool – it recommends five evaluation methods and analyzes them. These methods are: pros and cons analysis, Kepner-Tregoe (K-T) decision analysis, analytical hierarchy process (AHP), multi-attribute utility theory (MAUT), and cost-benefit analysis (CBA). Baker et al. (2001) state that these methods are adaptable to many situations, as determined by the complexity of the problem, needs of the customer, experience of the decision team/analysts/facilitators, and the time and resources available. No one decision-making method is appropriate for all decisions. Baker et al. (2001) present some hypothetical examples to facilitate understanding and use of these methods.

*Guidelines for Risk-Based Prioritization of DOE Activities*_DOE produced a standard for selecting or developing a risk-based prioritization (RBP) system, entitled 'Guidelines for Risk-Based Prioritization of DOE Activities', in April 1998. The standard describes issues that should be considered when comparing, selecting, or implementing a risk-based prioritization (RBP) systems. It also discusses characteristics that should be used in evaluating the quality of an RBP system and its associated results. DOE (1998) recommends the use of MAUT as an RBP model since it is a flexible, quantitative decision analysis technique and management tool for clearly documenting the advantages and disadvantages of policy choices in a structured framework. MAUT merits special consideration because it provides sound ways to combine quantitatively dissimilar measures of costs, risks, and benefits, along with decision-maker preferences, into high-level, aggregated measures that can be used to evaluate alternatives. MAUT allows full aggregation of performance measures into one single measure of value that can be used for ranking alternatives. However, DOE (1998) cautions that the results of MAUT analysis should not normally be used as the principal basis for decision-making. It will always be necessary to take into account factors that cannot be readily quantified, e.g. equity. Furthermore, it says that no technique can eliminate the need to rely heavily on sound knowledge, data, and judgments, or the need for a critical appraisal of results.

DOE used a multi-attribute model as the core of its Environmental Restoration Priority System (ERPS) developed in the late 80s (Jenni et al., 1995). Although ERPS was designed to operate with any specified set of values and trade-offs, it was used with values, including those based on risk analysis, which were elicited from DOE managers. This fact, coupled with the complexity of the model, gave it a lack of transparency that was troubling to stakeholders. In addition, the internal structure of

ERPS required that impacts be viewed in a trade-off analysis, which was equivalent to converting all effects into costs. This philosophical approach was particularly controversial with respect to regulatory compliance, where many observers noted that DOE must obey the letter of the law. DOE headquarters decided not to apply ERPS because of stakeholder opposition, although similar decision support systems have since been adopted for use at various DOE sites (CRESP 1999).

DOE has also attempted to use simple weighting to aid program planning and budget formulation processes (CRESP, 1999). For example, Risk Data Sheets focus on risk reduction but also recognize the need to address other program objectives (Compliance with pertinent laws and regulations; Mission impact; Mortgage reduction; and Avoidance of adverse social, cultural, and economic impacts).

3.4. EUROPEAN UNION

A detailed review of the regulatory background and use of decision analytical tools in European Union was recently conducted within the Contaminated Land Rehabilitation Network for Environmental Technologies (CLARINET, 2002) project. The review found that environmental risk assessment, cost-benefit analysis, life cycle assessment and multicriteria decision analysis were the principal analytical tools used to support environmental decision-making for contaminated land management in 16 EU countries (Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, United Kingdom). Similar to the US, quantitative methods like ERA and CBA are presently the dominant decision support approaches in use (e.g., SADA, WILMA) while MCDA and explicit tradeoffs are used less frequently (e.g., WEV).

Pereira and Quintana (2002) reviewed the evolution of decision support systems for environmental applications developed by the EU Joint Research Center (JRC). The concept of environmental decision support has evolved from highly technocratic systems aimed at improving understanding of technical issues by individual decision makers to a platform for helping all parties involved in a decision process engage in meaningful debate. Applications developed in the group include water resources management, siting of waste disposal plants, hazardous substance transportation, urban transportation, management, and groundwater management.

4. MCDA Application for Contaminated Land Management and Related Uses

4.1. MCDA APPLICATION TO MANAGEMENT OF CONTAMINATED SITES

Our review of the recent literature (published within the last 10 years), which is neither exhaustive nor complete, reveals only a few studies that utilize MCDA techniques to facilitate decision making for the management of contaminated sites. Table 1 summarizes the results of these studies. Most of these studies were conducted for the US Department of Energy.

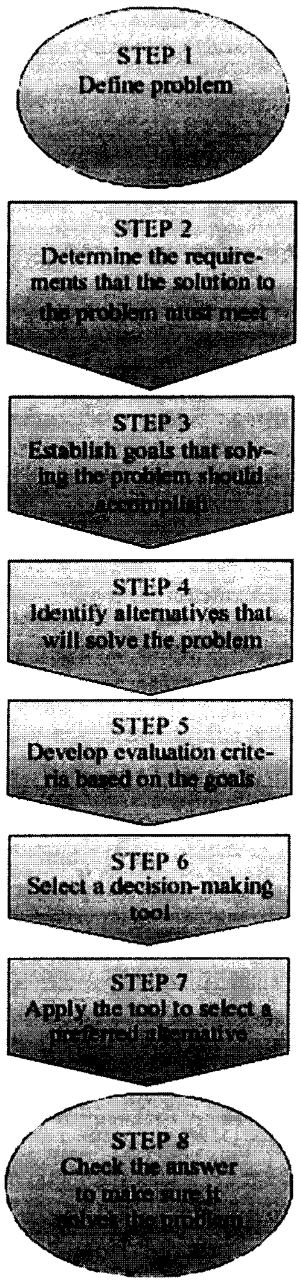

Figure 4: DOE General Decision Making Process.

DOE has sponsored a series of studies designed to develop decision models to analyze and select technologies for waste site remediation. Grelk (1997), Grelk et al. (1998) and Parnell et al. (2001) have developed a CERCLA-based decision analysis value model. The model incorporates five criteria: implementability, short-term effectiveness, long-term effectiveness, reduction of toxicity, mobility, or volume through treatment, and cost (Figure 5) that are further subdivided into a set of 21 measures. MAUT was used to determine weights associated with each individual measure. The model was used to perform analysis of remedial alternatives for a mixed-waste subsurface disposal site at Idaho National Environmental Engineering Laboratory (INEEL).

Ralston et al (1996) developed a generic model that incorporates life cycle cost and technological risk assessment for landfill waste site remediation. The model used MAUT to incorporate the decision maker's preferences for cost and time.

Timmerman et al. (1996) proposed the application of MAUT to DOE's problem of selecting the least risky technology to develop. Preferences of three end-users, expressed as MAUT utility functions, for two aspects of technical risks (the risk of successful development and the risk of successful implementation in the field) were assessed for several remediation processes. These utilities were then used to rank several treatment methods.

Deschaine et al., 1998 used a MCDA simulation model based on AHP to select the most promising remediation projects from a 114 radiological site remediation portfolio at the US Department of Energy's Savannah River Site. The model organizes the project's key quantitative (cost and savings) and qualitative (complexity of implementation, organizational importance, regulatory interface) information. The model is recommended for use in budget planning or resource allocation.

The studies reviewed above focused on evaluation of technical risk and comparison of alternative technologies; environmental risk assessment and stakeholder opinions were not quantified in these studies.

Apostolakis and his colleagues (Apostolakis, 2001, Bonano et al., 2000, Accorsi et al., 1999a & b) developed a methodology that uses AHP, Influence Diagrams, MAUT, and risk assessment techniques to integrate the results of advanced impact evaluation techniques with stakeholder preferences. In this approach, AHP is used to construct utility functions encompassing all the performance measures (criteria) and once created, MAUT is applied to compute expected utilities for alternatives. Accorsi et al. (1999a & b) used this approach to select a suitable technology for the cleanup of a contaminated site. The remedial action alternatives (RAAs) were ranked based on these expected values. A performance Index (PI) of the different alternatives was calculated.

Bonano et al., (2000) elicited stakeholder input for evaluating the impact of RAAs at a contaminated site. The authors incorporated fuzzy logic to rank criteria. Instead of using a single value, a triangular distribution for the AHP score was elicited. Apostolakis (2001) applied the methodology for choosing the best remedial action alternative for removing TCE and Chromium from a hazardous waste landfill.

4.2 MCDA APPLICATON TO GENERAL ENVIRONMENTAL MANAGEMENT

MCDA methods have been extensively applied to solve problems similar to those arising in the management of contaminated land. We summarize in this section decision analysis applications published in English language journals over the last 10 years that were located through Internet and library database searches. Each identified article was classified into one of five application areas: 1) prioritization of site/areas for industrial/military activity, 2) environmental/remedial technology selection, 3) environmental impact assessment, 4) stakeholder involvement, and 5) natural resource planning. If articles could potentially fit under multiple categories, the "prioritization of sites" category was preferentially emphasized. The articles and findings are summarized and presented in Table 2.

We should note that MCDA was also applied in other related areas. Keefer et al. (2002a, 2002b) conducted an exhaustive survey of decision analysis applications published in 1991-2001. They report the following application areas: Energy, Manufacturing and Services, Medical, Military, and Public Policy. We have also identified papers and books that review MCDA application in climate change (Bell et al., 2003), industrial facility sitting (Larichev and Olson, 2001), energy policy (Hobbs and Meier, 2000), agricultural resource management (Hayashi, 2000), and life cycle assessment (Seppala et al., 2002).

4.2.1. Prioritization of sites/areas for industrial/military activity Management of contaminated sites often requires site zoning for remediation, restoration or other uses. Even though we have not found any applications of MCDA methods for contaminated site zoning, all MCDA methods reviewed in this study (MAUT, AHP and outranking) have been used, in conjunction with GIS, for selection of the site boundaries and identification of geographical areas for related uses (e.g., industrial or military).

Mendoza et al. (2002) used AHP for allocating areas for military training exercises at Ford Hood, Texas. GIS spatial analysis techniques were used to generate average statistics for parameters associated with environmental conditions after training at each alternative training area (erosion status, land cover and land condition). The authors used an AHP-based framework to select the best training area depending on the intensity of training planned.

Keisler and Sundell (1997) proposed a generic framework that integrates MAUT and spatial analysis to determine National Park boundaries. It incorporates social development objectives and environmental preservation goals. The framework is

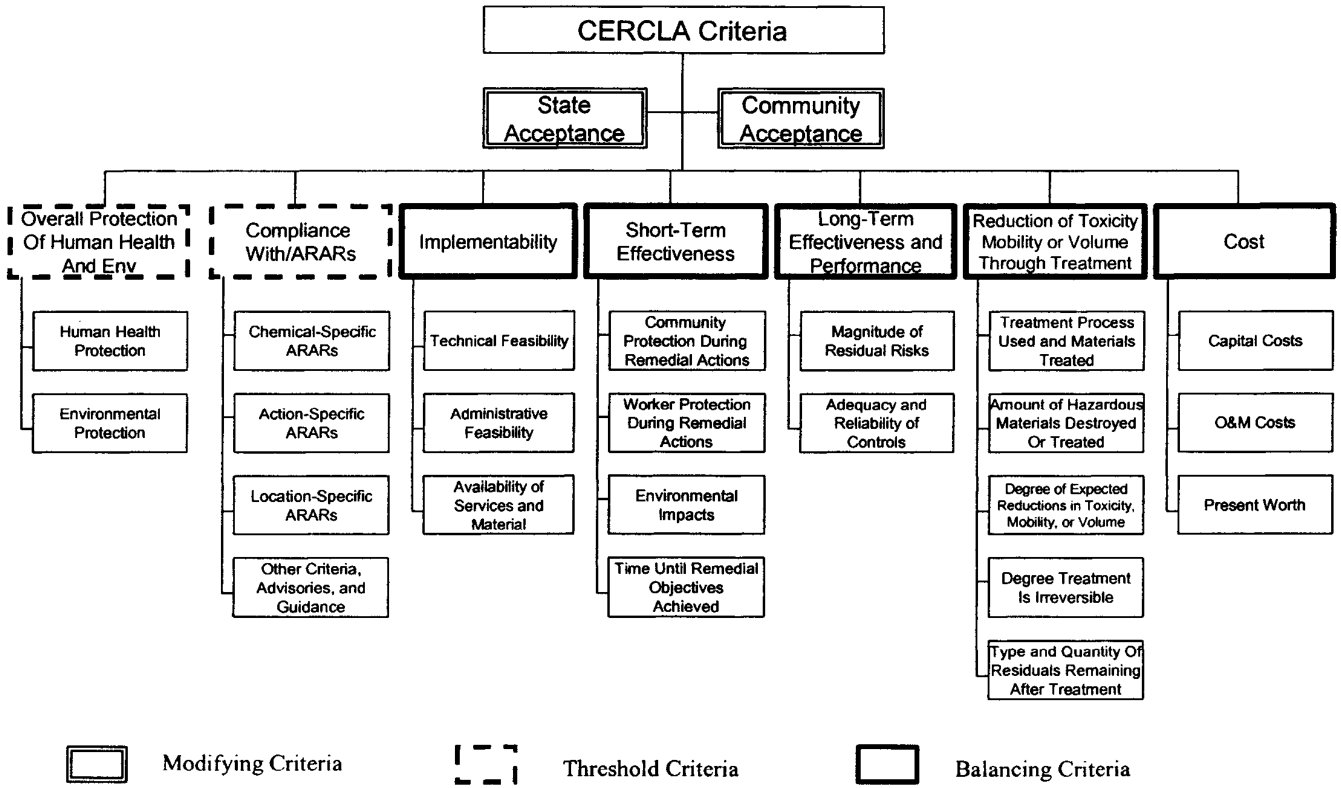

Figure 5. CERCLA Criteria Hierarchy

designed to help park managers and thus utilizes value objectives that are thought to be important for them. Sharifi et al., 2002 used MAUT-based analysis coupled with GIS to locate sustainable boundaries between a national park and Cochabamba City (Bolivia) to prevent unauthorized settlements. In this application, stakeholder (i.e. local population) pressure is most importance and the framework suggested in the study incorporates value judgments of the local population.

Joerin and Musy (2000) developed a generic method to integrate multiple considerations related to land management. They used GIS to develop a set of geographical descriptors and a MCDA-outranking method to aggregate this information and choose the optimal solution consistent with the preferences of decision-makers. They applied this methodology to develop a map of land suitability for housing in Switzerland. The following criteria were considered: impacts, air quality, noise, accessibility, climate, utility networks (e.g., water, electricity) and aesthetics.

Villancourt and Waaub (2002) used outranking and a GIS framework to select a site for a new waste management facility in Montreal.

4.2.2 Environmental/Remedial Technology Selection The selection of a feasible remedial action is usually the final stage of a contaminated site investigation (for example, it is required under CERCLA). Our review has identified several papers in which MCDA methods were utilized to select the best technology or remedial method. These studies cover all the MCDA techniques reviewed above.

A MAUT-based method was applied to compare current and alternative water control plans in the Missouri River (Prato, 2003). Structural modifications to the river have significantly altered its fish/wildlife habitat and thus resulted in the need for careful ecosystem management. The following criteria were considered: flood control, hydropower, recreation, navigation, water supply, fish and wildlife, interior drainage, groundwater, and historic properties. The analysis supported the implementation of a modified plan that incorporates adaptive management, increased drought conservation measures, and changes in dam releases. The decision framework implemented by Prato (2003) was tailored to the individual decision maker.

A related problem of regulating water flow in a river-lake system was addressed by Hamalainen et al. (2001) from the perspectives of group decision theory and stakeholder consensus building. The authors use MCDA-based approaches to incorporate stakeholder judgment during all stages of the project (problem structuring, identification of efficient alternatives, group consensus building, and public acceptance seeking).

Wakeman (2003) used the SMART technique to decide which action alternative to implement in handling the contaminated river sediment at Milltown Dam, MT. Factors considered in the study included availability of materials and services, ability to construct, and reliability.

One of the most advanced applications of MCDA techniques in this area was implemented for nuclear accident emergency management as a part of the EU-RODOS project (Ehrhardt and Shershakov, 1996). RODOS utilizes a MAUT analysis for strategy selection for population protection after a nuclear accident. Even though it's not a major consideration in the approach, terrestrial contamination is considered by RODOS software.Hamalainen et al., 2000 report the use of the software in a case study

of a simulated nuclear accident. The approach was found to be useful in explaining and justifying group decisions that needed to be made rapidly.

4.2.3 Environmental Impact Assessment Environmental impact assessments (EIA) are routinely conducted for all major projects with the potential to affect the environment (such as engineering/construction/industrial projects). The assessment of site contamination is often an integral part of EIA. Even though our review has not identified MCDA applications to EIAs specific to contaminated land, other applications are reviewed below.

Janssen (2001) reviewed 21 EIAs conducted in the Netherlands in the period of 1992-2000. The projects included construction (highways, railways, roads), river and freshwater reservoir development, and waste/sludge treatment plants, among others. Most of the EIAs reviewed used weighted summation methods, but a few projects used AHP and MAUT-based approaches.

Marttunen and Hamalainen (1995) reviewed methods for decision analysis interviews used in EIAs. MAUT/SMART and the AHP methodology were used and compared for the assessment of environmental impacts of a water development project in Finland. SMART was chosen over AHP because the AHP procedure proved to be too time consuming for stakeholders in a prior study.

Ramanathan (2001) recommended the use of AHP for considering multiple criteria and multiple stakeholders in EIA, and uses AHP to assess the socio-economic impact of a proposed Liquefied Petroleum Gas (LPG) recovery plant in an industrial area in India. The authorities responsible for the project wanted to know the *relative* importance of potential significant impacts, e.g. comparing the severity of impacts on housing with impacts on sanitation. AHP provided the methodology with which to capture perceptions of different people, and converting these perceptions to objective numbers in order to make a decision.

Rogers and Bruen (1998a) used ELECTRE III methodology in evaluating thresholds for noise impacts from a highway project in Ireland. Al-Rashdan et al., 1999 used PROMETHEE methodology to rank environmental impact assessments related to wastewater projects in Jordan. The methodology was found to be very useful in solving problems with conflicting criteria.

4.2.4 Natural Resource Management. Management of natural resources is the area with the most numerous examples of MCDA applications. Steiguer et al., 2003 developed an annotated bibliography that includes 124 important references ranging from theoretical studies to real-world applications of MCDA for forests and natural resource management. The authors argue that MCDA constitutes a newer and perhaps more acceptable method for quantifying and evaluating public preferences. Nevertheless, only a fraction of the reviewed studies included empirical testing of MCDA utility or feasibility and in most of the studies, researchers used hypothetical data or, at best, simplified decision situations; few studies were designed to implement a MCDA-generated management strategy.

The AHP within MCDA is receiving special attention for natural resource applications (Steiguer et al., 2003). Application of AHP in natural resource planning is summarized in the book by Schmoldt et al. (2001). The book illustrates the use of AHP

(with other analytical tools and extensions (e.g., fuzzy sets, GIS, MAUT) for forestry decision analysis, managing national park resources, participatory natural resources planning, forest industry investment strategies, salmon habitat restoration, biodiversity conservation, and assessing resource conditions in rural catchments.

Table 2 lists just a few representative publications. Schmoldt et al., 1994; and Schmoldt and Peterson, 2001b used AHP to address different aspects of natural park management, including developing inventory and monitoring programs as well as strategic management plans. Eight projects in Olympic National Park in Washington were prioritized with respect to implicit management objectives. The process was well received by the resource managers.

Pavlikakis and Tsihrintzis (2003) evaluated the utility of MAUT and AHP in selecting a technically suitable and socially acceptable management plan for national park in Eastern Macedonia and Thrace in Greece. The ranking of management alternatives received using AHP and MAUT methods were compared. Three methods consistently rank one of the four alternatives as less desirable. The ranks for three remaining alternatives were found to be inconsistent across the methods used.

MCDA methods have been extensively applied to a wide range of projects in forest management. AHP was applied for a project-scale forest management problem by Rauscher et al., 2000. MAUT analysis was applied to identify policy alternatives to manage a budworm outbreak in a local site in Canada (Levy et al., 2000). Kangas et al., 2001 tested application of several methods (MAUT and outranking) for large-scale forest policy planning in Finland. Even though most of these MCDA studies include social components, it can be used for technical evaluation alone. Store and Kangas (2001) used MAUT-based methods to conduct a habitat suitability evaluation over large areas. Finally, Tran et al., 2002, used AHP to assess environmental vulnerability across the Mid-Atlantic Region.

MCDA has also been applied to manage aquatic resources. Simon and Pascoe (1999) reviewed applications of MCDA in fisheries management. Brown et al. (2001) used weighting-based tradeoff analysis to select a management option for Buccoo Reef Marine Park in Tobago; criteria evaluated included ecological, social and economic factors. McDaniels (1995) used a MAUT approach to select among alternative commercial fishery openings involving conflicting long-term objectives for salmon management.

4.3 MCDA APPLICATON FOR STAKEHOLDER INVOLVEMENT AND CONSENSUS BUILDING

Examples presented above often attempted to visualize value judgment by a single decision maker and incorporate these value judgments to support the overall decision-making process. Stakeholder values are often considered as one attribute, along with others (such as costs, risk reduction, etc.) This section presents examples where decision-analytical procedures were used to incorporate and quantify stakeholders' values in cleanup and management alternatives using MCDA tools. MCDA can also be used as a framework that permits stakeholders to structure their thoughts about pros and cons of different remedial and environmental management options. MCDA

applications for group decision-making in other areas were also reviewed by Bose et al. (1997) and Matsatsinis and Samaras (2001).

Arvai and Gregory (2003) was the only identified study dealing with the application of decision-analytical tools to include stakeholder involvement at contaminated sites (Table 3). The authors compared two approaches for involving stakeholders in identifying radioactive waste cleanup priorities at DOE sites: (i) a traditional approach that involved communication of scientific information that is currently in use in many DOE, EPA and other federal programs, and (ii) a values-oriented communication approach that helped stakeholders in making difficult tradeoffs across technical and social concerns. The second approach has strong affinity to the MAUT-based tradeoffs discussed earlier in this chapter. The authors concluded that the incorporation of value-based tradeoffs information leads stakeholders to making more informed choices.

Table 3 summarizes several other representative stakeholder involvement studies in the areas related to management of natural resources and technology selection. We reviewed studies that address involvement of local communities at action specific levels, rather than broad-based public involvement efforts.

Several studies propose the use of MCDA tools for consensus building. Several papers by Gregory and McDaniels advocate the utility of this application and illustrate the use of value-oriented approaches that are based on MAUT. In general, applications may include individual surveys and workshops designed to elicit value judgment and construct decision alternatives. Specific applications include water resource management (McDaniels et al., 1999; Gregory et al., 2001); mining (Gregory and Keeny, 1994); wilderness preservation (McDaniels and Roessler; 1998), and estuary management (Gregory and Wellman, 2001). This study concludes that value-based approaches result in a higher level of comfort for participants and are useful in developing consensus-based management decisions. MAUT-based applications were also reportedly used in stakeholder value elicitation for regional forest planning (Ananda and Herath (2003), air quality valuation (Kwak et al., 2001), and agricultural applications (Gomez-Limon et al., 2003).

Schmoldt and Peterson (2001a) advocated the use of AHP as a decision support tool in workshop settings. They discussed an application of AHP in forest resource management.

The examples presented above used MCDA to facilitate consensus building. An alternative application of MCDA is in the organization of diverse interests, instead of seeking consensus-based middle ground. Gregory and Failing (2002) argue that a clear expression of difference facilitates development and acceptance of management plans. CMU approach to risk ranking (Morgan et al., 200, Florig et al., 2001) involves participants being elicited both as individuals and then in a group for i) holistic rankings of options that involve multiple objectives; and ii) individual attribute weights -- so that the two methods can be compared. Mental Modeling (Morgan et al., 2002) is a promising tool for assessing individual judgments. It involves individual, one-on-one interviews, leading participants through a jointly determined agenda of topics. The method allows free expression and encourages elaboration on topics in order to reveal individual perspectives at considerable depth. When done well, analysts can identify what people believe and why they believe it. They are also able to compare analyses

over time and provide insights into why beliefs may have changed. Environmental applications of mental modeling include management of the Illinois River basin in eastern Oklahoma (Focht et al., 1999, Whitaker and Focht, 2001), and in energy policy (Gregory et al., 2003).

5. Discussion

A decision-making process for contaminated land management must consider environmental, technological and social factors. Each of these factors includes multiple sub-criteria, which makes the process inherently multi objective.

Even though technical evaluations (such as risk assessment and feasibility studies) may be perceived as quantifiable and concrete, in reality uncertainty associated with these assessments may be very high. Scientists and engineers conducting remedial investigations and feasibility studies may attempt to provide an objective analysis of the competing remedial and abatement policies, but the decisions they make at each stage of the analysis (such as scoping out the problem, selecting a benchmark, developing appropriate models, etc.) can greatly affect the conclusions of the risk assessments and feasibility studies delivered to the decision makers (regulators and stakeholders). Most of these decisions are not well documented and not justified, which makes it difficult to assess the degree of conservatism (or lack of it) built into the analysis. Based on our experience in conducting and reviewing Superfund site risk assessments and our participation in model inter comparisons the uncertainty associated with typical models and parameters used in this process could result in an uncertainty range of several orders of magnitude in risk estimates (Linkov and Burmistrov, 2003).

Uncertain outputs from environmental and engineering evaluations are just one component of the problem that a decision maker may be concerned about. Environmental laws and regulations governing remedial/abatement processes implicitly and explicitly list multiple criteria that should be taken into account in addressing these issues. Most notably, CERCLA lists several categories of criteria (balancing, threshold and modifying) that should be met in the CERCLA process. It is not surprising that the practical decision making process does not incorporate these criteria explicitly, does not include transparency and explicit trade-offs and is often driven by the risk of litigation, rather than environmental risk.

Stakeholder involvement is a regulatory requirement for contaminated land management and almost every environmental project involves some sort of a public hearing. Elected representatives are often involved in some way (even if only to make funding decision), which is an example of an indirect public participation process. Stakeholder participation is typically limited in this setting and is not "value" driven. Consequently, current practice treats stakeholder participation as a constraint -- i.e., potentially controversial alternatives are eliminated early. Little effort is devoted to maximizing stakeholder satisfaction; instead the final decision is something that no one objects too strenuously to. Ultimately, this process does little to serve the needs or interests of the people who must live with the consequences of an environmental decision: the public.

TABLE 1. Applications of Decision Support Tools for Contaminated Sites

Method	Site Type	Decision Context	Criteria	Funding Agency	Citation
AHP + MAUT+Fuzzy	Hazardous chemical waste landfill located at a US National Laboratory	Selection of Remedial Action Alternative	Programmatic, Life Cycle cost, Socioeconomic Impact, Environmental, Human Health and Safety	DOE	Accorsi, 1999a, 1999b, Bonano, 2000, Apostolakis, 2001
AHP + Linear Programming	Savannah River Site (nuclear and chemical industries)	Optimal budget/resource allocation for remediation	Cost of remediation, mortgage cost reduction, technical feasibility, annual funding constraints	DOE	Deschaine, 1998
MAUT	INEEL subsurface disposal area	Selection of Remedial Action Alternative	implementability, Short-term effectiveness, long-term effectiveness, reduction of toxicity/mobility/volume, cost	DOE	Grelk, 1997, Parnell et al., 2002
MAUT	INEEL landfill	Selection of Remedial technology	Cost, Time & Risk	DOE	Ralston et al., 1996
MAUT	INEEL landfill	Selection of Remedial technology	Risk of successful development and Risk of successful field implementation	DOE	Timmerman et al., 1996
AHP	Contaminated DoD Installations in Korea	Selection of Brownfield management technologies	Resource requirements, data quality, method limitations, compliance with policy, input requirements, and output	DoD	Hartman and Goltz, 2002

TABLE 2. Applications of Decision Support Tools in Areas Related to Contaminated Site Management

Application Area	Method	Decision Context	Funding Agency	Citation
Prioritization of sites/areas for industrial/military activity	AHP + GIS	Land Condition Assessment for allocation of military training areas	US Army Engineering Research and Development Center	Mendoza et al., 2002
	AHP+GIS	Selection of Boundaries for National Park	International Institute for Geoinformation Science and Earth Observation, Netherlands	Sharifi et al., 2002
	PROMETHEE	Waste management activities in Canada	Natural Sciences and Engineering Research Council of Canada	Vaillancourt et al., 2002
	ELECTRE + GIS	Land Management: develop a land suitability map for housing in Switzerland	Swiss National Foundation for Research (FNRS)	Joerin et al., 2000
	AHP + GIS	Landfill sitting		Siddiqui et al., 1996
	MAUT + GIS	Selection of Park Boundaries	DOE	Keisler et al., 1997
Environmental/Remedial Technology Selection	SMART	Choosing a remedial action alternative at Superfund Site	US Army Corps of Engineers	Wakeman, 2003
	MAUT	Selection of management alternative Missouri River	University of Missouri-Columbia	Prato, 2003
	MAUT+AHP	Regulation of water flow in a Lake-River system	Academy of Finland	Hamalainen et al., 2001
	MAUT	Offsite emergency management following a nuclear accident (such as the Chernobyl accident)	European Commission, Ukraine	Ehrhardt, 1996; Hamalainen et al., 2000
Environmental Impact Assessment	Review	Review of MCDA use for EIAs in Netherlands	Vrije University, Netherlands	Janssen, 2001
	AHP	Socio-economic impact assessment for a construction project in India	Indira Gandhi Institute of Development Research	Ramanathan, 2001
	ELECTRE	Highway environmental appraisal in Ireland	Dublin Institute of Technology; University College Dublin	Rogers et al., 1998a
	AHP & MAUT/SMART	Environmental Impact Assessment of 2 water development projects on a Finnish river	Finnish Environmental Agency; Helsinki University of Technology	Marttunen et al., 1995
	PROMETHEE	Prioritization of EIAs in Jordan	Staffordshire University, UK	Al-Rashdan et al., 1999

Natural resource Planning	AHP	Natural Park Management	USDA Forest Services	Schmoldt et al., 1994; Peterson et al., 1994; Schmoldt and Peterson, 2001
	AHP	Management of small forest in NC, USA	USDA Forest Services	Rauscher et al., 2000
	MAUT	Management of spruce budworm in Canadian forests	National Science and Engineering Research Council of Canada	Levy et al., 2000
	AHP, MAUT & Outranking	Forestry Planning in Finland	Finnish Academy of Sciences; Finnish Forest Research Institute	Kangas et al., 2001
	MAUT	Improvement of Habitat Suitability Measurements	Finnish Forest Research Instyitute	Store and Kangas, 2001
	AHP	Environmental Vulnerability assessment for Mid-Atlantic Region	US EPA/ DOE	Thran et al., 2002
	Weighting	Management of marine protected areas in Tobago	UK Department of International Development	Brown et al., 2001
	MAUT	Fisheries Management: select among alternative commercial fishery opening days	Fisheries and Ocean Canada	McDaniels, 1995
	AHP, MAUT & Outranking	Fisheries management		Mardle and Pascoe, 1999
Stakeholder Involvement	MAUT	Risk attitudes by farmers in Spain	EU; Spanish Government	Gomez-Limon et al., 2003
	MAUT	Air-Quality Valuation in Korea	Korea University	Kwak et al., 2001
	PROMETHEE	Sustaining exploitation of Renewable Energy Sources	National Technical University of Athens	Georgopoulou et al., 1998
	MAUT	Elicitation of Wilderness Preservation benefits	Social Sciences and Humanities Research Council of Canada; National Science Foundation	McDaniels et al., 1998
	MAVT	Regional Forest Planning	La Trobe University, Australia	Ananda et al., 2003

Note: If a funding agency wasn't stated in the paper, the authors' affiliation is given in the Funding Agency column.

TABLE 3. Applications of Decision Support Tools for Stakeholder Involvement

Method	Application Format	Decision Context	Funding Agency	Citation
MAUT	individual surveys performed under supervision	Generic radiologically contaminated site	DOE/NSF	Arvai and Gregory, 2003
MAUT	workshop and individual interviews	Consensus building for Water Resource Management	Social Sciences and Humanities Research Council of Canada; National Science Foundation	McDaniels et al., 1999; Gregory et al., 2001
MAUT	workshop	developing Alternatives for Coal mine exploration in Malaysia	Social Sciences and Humanities Research Council of Canada; National Science Foundation	Gregory and Keeney, 1994
MAUT	value elicitation in workshop settings	Elicitation of Value Judgements for Wilderness Preservation	Social Sciences and Humanities Research Council of Canada; National Science Foundation	McDaniels et al., 1998
MAUT	workshop	developing Alternatives for Coal mine exploration in Malaysia	NSF, EPA	Gregory and Wellman, 2001
MAVT	individual surveys	Regional Forest Planning	La Trobe University, Australia	Ananda et al., 2003
MAUT	individual surveys	Air-Quality Valuation in Korea	Korea University	Kwak et al., 2001
MAUT	individual surveys	Risk attitudes by farmers in Spain	EU; Spanish Government	Gomez-Limon et al., 2003
AHP	workshop	forest fire management	USDA	Schmoldt and Peterson, 2001
MAUT	individual surveys	water use planning		Gregory and Failing, 2002
Mental modeling	individual surveys, workshop	watershed management	EPA	Focht, 1999, Whitaker and Focht, 2001
Mental modeling	individual surveys, workshop	energy policy		Gregory, 2003

Note: If a funding agency wasn't stated in the paper, the authors' affiliation is given in the Funding Agency column.

The increasing volume of often controversial information being generated to support environmental management, and the limited capacity of individual decision makers to integrate and process that information, emphasizes the need for developing tractable methods for aggregating the information in a manner consistent with decision maker's values. The field of MCDA has developed methods and tools that can help in developing a decision analytical framework useful in the management of contaminated land. The purpose of MCDA is not to single out the correct decision, but to help improve understanding in a way that facilitates a decision making process involving risk, multiple criteria and conflicting interests. MCDA visualizes tradeoffs among multiple conflicting criteria and quantifies uncertainties necessary for comparison of available remedial and abatement alternatives. This process helps technical project personnel as well as decision makers and stakeholders to systematically consider and apply value judgments to derive an optimal policy alternative. MCDA also provides methods for participatory decision-making where stakeholder values are elicited and explicitly incorporated into the decision process.

MCDA methods have there associated strengths and limitations. Although easy in implement, the Elementary Methods reduce complex problems to a singular metric and thus can result in an oversimplified and often overly conservative representation of the problem. MAUT is one of the most scientifically grounded methods with a strong foundation in decision theory. It requires developing a utility function and making difficult tradeoffs, a time consuming and sometimes frustrating process for decision makers and stakeholders that limits MAUT application. Comparisons and derivation of weights are simplified in AHP-based approaches, but the theoretical foundation for its computational algorithms as well as its inherent linearity has made AHP the subject of intense controversy. Outranking takes a different approach – eschewing compensatory optimization functions and introducing greater flexibility by allowing semi-quantitative scales. Selecting a method from the methods available may be itself an expression of subjective values (such as a preference for non, partially, or completely compensatory approaches), or a purely pragmatic choice (such as familiarity or perceived ease of implementation).

No matter which analytical decision tool is selected, implementation requires complex, often impossible, tradeoffs. This complexity is probably one of the main reasons why MCDA is still not widely used in practical applications. However, explicit, structured approaches will often result in a more efficient and effective decision process compared with the often intuitive and biased processes that are currently in place.

Formal applications of MCDA in management of contaminated sites are still rare. Applications in related areas are more numerous, but to-date they have remained largely academic exercises. One exception is the use of AHP-based methods in natural resources planning. Nevertheless, the positive results reported in the studies reviewed in this paper as well as the availability of recently developed software tools provides more than an adequate basis for recommending the use of MCDA in contaminated site management.

6. Proposed Framework

Here we provide an overview of the systematic decision analytic framework that incorporates the use of MCDA. We use the term decision analysis to refer to a set of quantitative methods for analyzing decisions, rather than a description of how unaided decisions are made. The proposed framework is illustrated Figure 6 and is intended to give a generalized "road-map" to the decision process. Figure 6 acknowledges the fundamental roles of "people, process and tools" within complex, environmental decisions. The application of these three components depends on the scale and extent of the decision challenge.

The "process" section, shown in Figure 6, is central to the overall decision in providing an adaptable structure so that participants can modify aspects of the project to suit local concerns, while still producing the required outcomes. The process follows two basic themes, (1) generating alternatives, criteria and values and (2) ranking the alternatives by applying the user-defined criteria and values. As with most other decision processes reviewed in this paper, it is assumed that the process in Figure 6 is iterative at each phase and can be cycled through many times in the course of complex decision making. The same basic "process" is used initially with rough estimates to sketch out the basic elements and challenges in the decision process. As these challenges become more apparent one iterates again through the framework to explore and adapt the process to address the more subtle aspects of the decision, such as the need for detailed scientific studies or weighting stakeholder preferences.

Figure 6 also describes the contributions of "people" to the overall decision. The activity/involvement levels of three basic groups of people (decision-makers, scientists/engineers and stakeholders) are symbolized by dark lines for direct involvement and dotted lines for less-direct involvement. While the actual membership and the function of these three base groups may intersect or vary, the roles of each are essential in gathering the most utility from human input to the decision process. Each group has its own way of looking at the world, its own method of envisioning solutions and its own societal responsibility. They rely on each other to provide useful information and experience to the decision process.

As shown in Figure 6, the tools used within group decision-making and scientific research are essential elements of the overall decision. As with "people", the applicability of the tools is symbolized by solid lines (direct/high utility) and dotted lines (indirect/lower utility). Decision analysis tools help to generate and map group stakeholder preferences as well as individual value judgments into organized structures that can be linked with the technical inputs from risk analysis, modeling/monitoring, and cost estimations. The pruning part of the process entails the methodical and structured narrowing of non-feasible options by first screening mechanisms (for example, overall cost, technical feasibility, general societal acceptance) and then a more detailed ranking of the remaining options by subtler and finely tuned analytical techniques (AHP, MAUT, Outranking). The tools also provide useful graphical techniques and visualization methods to express the gathered information in understandable formats. When changes occur in the requirements or decision process, these tools can respond efficiently to reprocess and iterate with the new inputs.

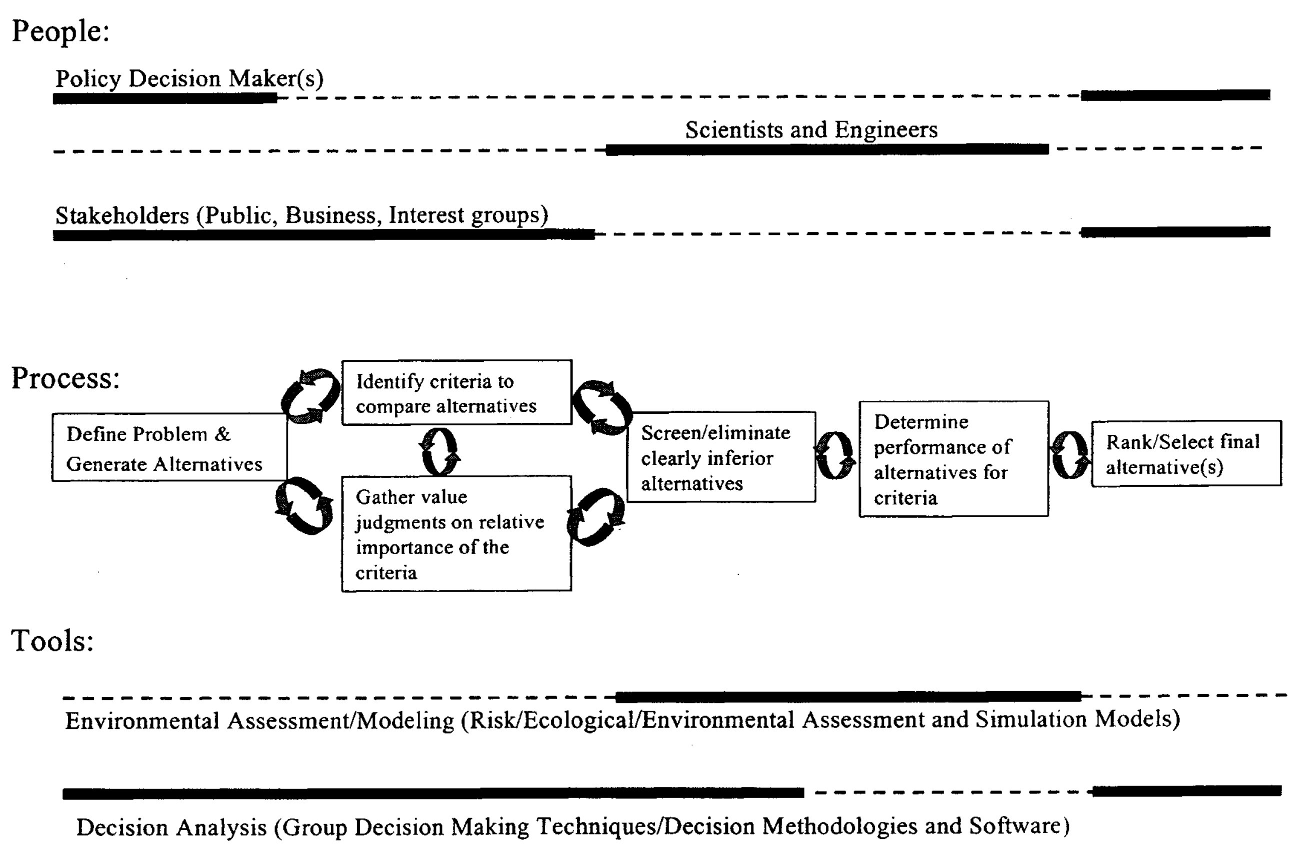

Figure 6: Proposed Framework

7. Conclusion

Environmental decision-making involves complex trade-offs between divergent criteria. The traditional approach to environmental decision-making involves valuing these multiple criteria in a common unit, usually money, and thereafter performing standard mathematical optimization procedures. Extensive scientific research in the area of decision analysis has exposed many weaknesses in this approach. At the same time, new methods that facilitate a more rigorous analysis of multiple criteria have been developed. These methods, collectively known as MCDA methods, are increasingly being adopted in environmental decision-making. This paper surveyed the principal MCDA methods currently in use and cited numerous environmental applications of these methods. While MCDA offers demonstrable advantages, choosing among MCDA methods is a complex task. Each method has strengths and weaknesses; while some methods are better grounded in mathematical theory, others may be easier to implement. Data availability may also act as a constraint on applicable methods. It is therefore unavoidable that the decision-maker will have to choose, on a case-by-case basis, the most suitable MCDA technique applicable to each situation. This paper has set out a decision analytic framework to facilitate such a selection process and thereafter provides guidance on the implementation of the principal MCDA methods. While the guiding principles provided here are expected to be useful, situation-specific expert judgment must inform the choice and implementation of decision analytic methods.

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Part 1

Comparative Risk Assessment: Methods, Tools and Applications

USING COMPARATIVE EXPOSURE ANALSIS TO VALIDATE LOW-DOSE HUMAN HEALTH RISK ASSESSMENT: THE CASE OF PERCHLORATE

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Abstract

Comparative risk assessment is usually performed to inform risk ranking and prioritization exercises. Here it is applied as an innovative tool for testing the scientific validity and reliability of a 2002 USEPA human health risk assessment of perchlorate. Dietary exposure to nitrate is compared with drinking water exposure to perchlorate; both chemicals act on the thyroid gland by iodide uptake inhibition (IUI). The analysis shows that dietary nitrate is predicted to cause orders of magnitude more IUI than perchlorate exposure at environmental concentrations. If the 2002 USEPA risk assessment is scientifically valid and reliable, then a generally accepted decade-old USEPA nitrate risk assessment is fatally flawed, and risk management decisions based on it are severely under-protective. If the nitrate risk assessment is valid and reliable, however, then the 2002 USEPA perchlorate risk assessment is fatally flawed, unreliable and should not be used as the basis for risk management. The origin of this inconsistency is a policy decision to deem IUI a "key event" that may lead to changes in thyroid hormones and consequent adverse effects. This implicitly treats IUI as "adverse." Unless large and sustained over a long period, however, IUI is mundane, reversible, and arises at exposure levels orders of magnitude below true adverse effects. In communities where quantitative human health risk assessment is expensive or expertise is lacking, comparative exposure assessment provides a cost-effective means to evaluate the merits of such assessments before taking costly risk management actions.

1. Introduction

Perchlorate anion (ClO_4^-) has been discovered in drinking water supplies at numerous locations in the US. In January 2002, the US Environmental Protection Agency issued an external review draft risk assessment proposing an oral reference dose (RfD) of 0.00003 mg/kg-day corresponding to a drinking water equivalent level of 1 ppb perchlorate ($DWEL_P$) [1]. These values were derived from animal data based on potential adverse effects on thyroid hormone production due to IUI at the sodium-iodide symporter (NIS), which it is suggested might lead to neurodevelopmental deficits in the fe-

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tus. A composite uncertainty factor of 300 is embedded in the RfD derivation. USEPA proposes that IUI be termed a "key event" that can lead to subsequent declines in thyroid hormone levels, which then "can lead to permanent neurodevelopmental deficits" ([1] at E-8). USEPA further suggests that changes in IUI may cause increased thyroid stimulating hormone (TSH); TSH changes may fail to compensate for decrements in thyroid hormone levels; and reduced thyroid hormone levels can cause changes in thyroid histopathology. USEPA proposed an assessment model that used changes in thyroid hormone levels as the "precursor lesions" of "subsequent effects that potentially could lead to thyroid tumors or to altered neurodevelopment" (Id.)

USEPA's 2002 draft perchlorate risk assessment is significant in at least three ways in relation to the Agency's overall risk assessment process. First, it explicitly implements the intent to harmonize cancer and noncancer approaches to risk assessment. To achieve this harmonization it proposes to use IUI as the single identifiable event that precedes both cancer and noncancer effects. Thus, it is a harbinger of the future for USEPA harmonized risk assessments. Second, it would extend the term "key event" to the threshold of a biochemical phenomenon—IUI—that is mundane, reversible and undetectable by the individual. Risk assessment practice would be shifted to estimating doses or exposures low enough to prevent "key events" rather than adverse effects. Third, the assumptions made in this risk assessment imply that USEPA's nitrate risk assessment is highly inaccurate and warrants early revision. They also implies that existing risk management policies based on the nitrate risk assessment are severely underprotective, and highlight the inconsistency between past and future USEPA's risk assessment practices.

Assessments of low-level chemical risks often are highly uncertain and controversial because model predictions cannot be verified or validated in accordance with traditional scientific methods. Predictions are made by extrapolating from exposure ranges where scientific knowledge exists to a lower exposure range where data cannot be observed. The inability to discover data directly applicable to low-dose risks is generally unavoidable and does not eliminate the need to make public policy decisions where science is lacking. This may be the single greatest source of controversy in risk analysis: The ability to test and refute predictions is the lynchpin of science, but the predictions of risk assessment models are generally not testable.

In the case of perchlorate, comparative exposure analysis provides an innovative way to evaluate (though not directly test) the validity and reliability of inferences from a human health risk assessment. Perchlorate and nitrate have very similar mechanisms of action on the thyroid gland but their potencies and exposure levels differ. Adjusting for these differences enables comparisons to be made in the relative magnitude of IUI. In this analysis, a Perchlorate Equivalency Ratio ('PER') is calculated as the relative potency of perchlorate to nitrate per unit of mass. Values as low as 10 and as high as 1,000 are derived from the literature and propagated through the analysis along with a conservative, best professional judgment (BPJ) estimate of 300. Intended precision in PER values is $\pm$ 0.5 $\log_{10}$. This uncertainty exceeds all others in the analysis, and despite its magnitude does not significantly affect inferences and conclusions.

The comparative exposure assessment predicts that single daily servings of common foods containing nitrate cause IUI at levels tens, hundreds or even thousands of times greater than that assumed to be caused by perchlorate at environmental expo-

sure levels. Milk and processed meats tend to be at the low end of this range, and leafy green vegetables occupy the high end. Because many of these vegetables are consumed raw, preparation method is not a significant potential confounder.

If USEPA's 2002 draft perchlorate risk assessment is valid and reliable and environmental exposure levels are deemed to pose a human health threat, then routine vegetable consumption must be a threat that is orders of magnitude greater. This would imply that USEPA's nitrate risk assessment [2] is seriously flawed because effects on the thyroid and subsequent sequelae are not addressed in the nitrate risk assessment. Further, USEPA's promulgated nitrate Maximum Contaminant Level (MCL) based on this risk assessment would be severely under-protective. For example, under the BPJ scenario described here, nitrate exposure of 300 ppb has the same IUI potential as perchlorate exposure at 1 ppb. This implies that the current nitrate MCL of 10,000 ppb is at least 30 times too high to achieve the same level of protection. If the nitrate risk assessment is reevaluated to address IUI, a precautionary risk management approach would likely rely on a lower PER than 300, such as 100 or 30. In that case, the nitrate MCL would have to be reduced to 100 or 30 ppb to provide the same level of protection as USEPA's 2002 proposed perchlorate RfD and 1 ppb $DWEL_P$. Note that the 1 ppb $DWEL_P$ based on USEPA's 2002 draft risk assessment is being used by both USEPA and various state regulatory agencies as an implicit drinking water standard.

However, if USEPA's 1991 final nitrate risk assessment is valid and reliable and standards based on it are in fact protective, then USEPA's 2002 draft perchlorate risk assessment is fatally flawed. In combination with accepted pharmacologic knowledge, the 2002 draft perchlorate risk assessment implies that routine nitrate exposure below the MCL poses a much greater threat to thyroid health than environmental perchlorate exposure.

2. The Thyroid and Sodium-Iodide Symporter

The primary responsibility of the thyroid is to produce the thyroid hormones thyroxine (T4) and triiodothyronine (T3). These hormones are essential for many aspects of normal function, including metabolism, growth, and reproduction. They are also critical for normal fetal development. These hormones are formed in the thyroid follicle cell (the functional unit of the thyroid) via organification of iodide. For this process to take place, iodide must be transported from circulation in the bloodstream into the thyroid follicle cell. This process involves a cell membrane bound protein called the sodium-iodide symporter, or NIS [4].

Perchlorate and nitrate are competitive inhibitors of the thyroidal uptake of iodide by the NIS [4, 5], and both ions act to prevent the uptake of iodide at the NIS. While these mechanisms are slightly different, the biochemical difference is expected to be insignificant as far as the effect on iodide uptake is concerned.

Homeostatic mechanisms mitigate any changes in thyroidal iodide uptake; the success of these mechanisms in maintaining normal thyroid function depends upon the duration and magnitude of IUI. The physiological consequences of a given degree of IUI are determined by intrinsic thyroid function, iodine nutrition status, species-specific factors, and the pattern and duration of exposure to the inhibitor.

If inhibition of thyroidal iodide uptake is great enough and prolonged enough to overcome the homeostatic regulatory mechanism, formation of thyroid hormone is reduced [13]. If thyroid hormone levels are reduced below the range compatible with normal thyroid function, then the reduction is physiologically significant and should be considered adverse. Substantial and sustained IUI is a precursor of physiologically significant reductions in thyroid hormone formation through this mode of action.

3. Goitrogens in Common Foods

An extensive review of the toxicological, medical, and environmental literature shows that multiple chemicals affect the thyroid system and can be found in common foods. These goitrogens include nitrate (which, like perchlorate, inhibits iodide uptake), thiocyanates (which inhibit iodine concentration), and isoflavones (which inhibit thyroid peroxidase). Of these goitrogens, nitrates provide the best analog for perchlorate because the mode of action is very similar. Nevertheless, other modes of action are sufficiently close that a more complete analysis is necessary to estimate the total potential goitrogenic effects of common foods relative to perchlorate. Table 1 lists foods and beverages known to contain anti-thyroid chemicals along with their mode of action. This list includes common root vegetables (*e.g.*, beets, carrots, and turnips), cruciferous vegetables (*e.g.*, broccoli, cauliflower) and legumes of many types, cured meats and milk. Some foods, such as milk and broccoli, have more than one goitrogen.

4. Data and Methods

Nitrate intake from common foods was converted to perchlorate dose-equivalents based on the relative potency of perchlorate to nitrate in terms of IUI. These perchlorate dose-equivalents are then compared to the dose of perchlorate that would be ingested daily assuming a drinking water concentration equal to USEPA's proposed 1 ppb $DWEL_P$. Because pregnant women are presumed to be the sensitive subpopulation for exposure, USEPA's proposed perchlorate RfD was converted into a daily dose of 0.002 mg based on a 64.2 kg reference woman consuming two liters of tap water per day [2].

Perchlorate dose equivalents were calculated for each food item according to the following model:

$$V = (N \times S \times B)/PER \text{ , where:}$$

V ≡ perchlorate dose-equivalent (mg);
N ≡ average nitrate concentration, in mg/kg;
S ≡ serving size, in kg;
B ≡ % nitrate bioavailability after gastrointestinal absorption; and
'PER' ≡ Perchlorate Equivalency Ratio, defined as the relative potency of nitrate to perchlorate on IUI per unit mass.

Data on nitrate concentrations (N) in common foods are provided in Table 2 [14]. Standard serving sizes (S) were obtained [15].

TABLE 1: Examples of Goitrogens in Common Foods

Anti-thyroid Chemical	Mode of Action	Examples of Common Foods
Nitrate	Inhibit iodide uptake	Milk Processed Meats Bacon, corned beef, pickled beef, ham, pepperoni (beef), sausage (various) Vegetables Artichoke, asparagus, bean (green), bean (lima), beet, broccoli, Brussels sprouts, cabbage, carrot, cauliflower, celery, corn, lettuce, radish, rhubarb, spinach, tomato, turnip, turnip greens
Thiocyanates and related compounds	Inhibit iodide concentration	Milk Broccoli
Disulfides	Inhibit iodide uptake and organification	Garlic, onion
Isoflavones	Inhibit thyroid peroxidase	Legumes Soy Soy-based infant formulas, soy-based meat substitutes, tofu Teas Green, jasmine, Lapacho
Goitren	Interferes with iodide organification	Milk Vegetables Bamboo shoots, bean (lima), brassica seeds, cabbage, cassava, maize, rutabaga, sweet potatoes
Compiled from [6-12].		

4.1. NITRATE BIOAVAILABILITY (B)

Chemicals with similar or even identical mechanisms of action may have very different biological effects if they vary in the extent to which they are bioavailable upon exposure. There is substantial empirical literature concerning perchlorate but relatively little about nitrate. However, the literature available suggests that perchlorate and nitrate do not differ appreciably in bioavailability.

Lambers *et al.* [16] performed an experiment on 12 persons using three nitrate-rich vegetables (plus a control of nitrate administered intravenously) to determine the extent to which indigested nitrate in vegetables is absorbed from the gastrointestinal

tract. After control for endogenous nitrate production, these percentages were 98% ± 12% for spinach, 113% ± 14% for lettuce, and 106% ± 15% for beetroot. Absent this correction, percentages were 91% ± 10% (spinach), 89% ± 13% (lettuce), and 106% ± 15% (beetroot). The consistency of these values provides confidence that nitrate bioavailability from vegetables is high and that a default assumption can be reliably applied to all foods. A value of 90% was used, and is expected to be conservative.

TABLE 2. Nitrate Levels in Common Foods [14, 18]

Vegetables	Concentration (mg/kg)	Meats and Dairy	Concentration (mg/kg) (mg/L)
Artichoke	12	Bacon (unsmoked side)	134
Asparagus	44	Bacon (unsmoked back)	160
Beans (green)	340	Bacon (peameal)	16
Beans (lima)	54	Bacon (smoked)	52
Beets	2,400	Beef (corned)	141
Broccoli	740	Beef (cured corned)	852
Brussels sprouts	120	Beef (corned brisket)	90
Cabbage	520	Beef (pickled)	70
Carrots	200	Beef (canned corned)	77
Cauliflower	480	Ham	105
Celery	2,300	Ham (smoked)	138
Corn	45	Ham (cured)	767
Cucumber	110	Pepperoni (beef)	149
Eggplant	270	Sausage (summer)	135
Endive	1,300	Sausage (Ukrainian Polish)	77
Kale/collard	800	Sausage (German)	71
Leek	510	Milk	5
Lettuce	1,700		
Melon	360		
Mushroom	160		
Onion	170		
Parsley	1,010		
Peas	28		
Sweet pepper	120		
White potatoes	110		
Sweet potatoes	46		
Pumpkin	400		
Squash	400		
Kimchi	1.6		
Radish	1,900		
Rhubarb	2,100		
Spinach	1,800		
Tomatoes	58		
Turnip	390		
Turnip greens	6,600		

4.2 'PER' DERIVATIONS

The Perchlorate Effectiveness Ratio ('PER') is defined as the ratio of the mass of nitrate to the mass of perchlorate needed to achieve a particular thyroid effect. For example, if a drinking water study in rats found that 20 mg perchlorate resulted in a decrease of 50% in IUI, and 200 mg nitrate had the same effect, the nitrate PER for that study would be 200/20 = 10.

Several studies in the pharmacology literature directly compare the effectiveness of perchlorate on IUI relative to other chemicals. Greer *et al.* [17] determined relative potencies by incubating sheep thyroid slices with radioiodine and various concentrations of iodine uptake inhibitors, including nitrate and perchlorate. The authors estimated that the 50% inhibitory concentrations of perchlorate and nitrate were 24 μmol and 10 mmol, respectively. On a weight-adjusted basis, these correspond to 2.4 mg of perchlorate and 620 mg of nitrate, for a nitrate PER of 260.

Wyngaarden *et al.* [19] studied the effect of interperitoneally injected doses of various chemicals on radioiodine uptake in the rat thyroid. An estimate of the relative potencies of perchlorate and nitrate were obtained from the dose-response curves for the two chemicals displayed in their Figure 1. Doses that cause 20% inhibition (a ratio of 80 on the *y*-axis in their Figure 1) were identified because relatively low doses of thyroid-affecting chemicals are of interest for this comparison. Using this method, the weight-adjusted doses at 20% inhibition for nitrate and perchlorate are 60 μg and 7 μg, respectively, for a nitrate PER of 9.

Wyngaarden *et al.* [20] used a similar method to Wyngaarden *et al.* [19], but instead measured uptake of radioiodine in rats using a neck radiation detector instead of analyzing compounds in serum and thyroid. The authors do not report their actual data but state: "A dose-response study revealed perchlorate to be 10 times and nitrate 1/30 as potent as thiocyanate in discharging iodide previously collected by the thyroid... The capacity of these agents to prevent the collection of iodide by the thyroid approximately paralleled their iodide discharging action." Based on this study, the nitrate PER would be 300.

Limited evidence and differences in methods indicate substantial uncertainty concerning the true nitrate PER. We believe that a range of 10 to 1,000 captures this scientific uncertainty, which we propagate throughout the analysis that follows. A value of 10 would predict that nitrate and perchlorate have similar potencies, whereas a value of 1,000 would predict that perchlorate is much more potent. Our conservative, best professional judgment (BPJ) nitrate PER is 300. In logs it is closer to 1,000 than 10. We suspect that 10 may be too low (though it is about the lowest ratio derived from any study) and think that 1,000 is too high (it exceeds by 3*x* the highest ratio derived from any study).

There is no direct evidence that similar results would also be observed in humans. However, USEPA's 2002 draft risk assessment presumes that thyroid effects in rats may be reliably extrapolated to humans. Also, there is evidence that the NIS is conserved across species [21]. With respect to other factors of interest, Greer *et al.* [22] found no difference in iodide uptake between human males and females in a clinical trial which estimated the no-effect level threshold for IUI at 180-220 ppb.

4.3 NITRATE DRINKING WATER STANDARD

USEPA has issued a national primary drinking water standard ("Maximum Contaminant Level", or MCL) of 10 ppm for nitrate based on methemoglobinemia as the critical effect [2]. Derivation of the MCL did not take into account the potential effect of nitrate on IUI or other thyroid endpoints. An additional comparison could be made based on the ratio of the nitrate MCL (10 ppm) to the implied perchlorate DWEL (1 ppb), yielding a $MCL_N/DWEL_P$ PER of 10,000. MCLs are set at levels to protect human health taking account the science, technical feasibility and cost, whereas RfDs presumably do not account for technical feasibility and cost. Therefore, a PER based on $MCL_N/DWEL_P$ is not science-based. However, the 1 ppb $DWEL_P$ based on USEPA's 2002 draft risk assessment is being used by as an implicit drinking water standard, weakening the normal distinction between RfDs (ostensibly scientific values) and MCLs (unambiguously regulatory standards). Also, as a risk communication matter this PER would be implied by their co-existence and simultaneous use. The comparison is relevant to the extent that USEPA's perchlorate risk assessment implies error in both its previous nitrate risk assessment and the MCL based largely upon it.

5. Results

5.1 ALTERNATIVE SCIENCE-BASED 'PER' SCENARIOS

Although perchlorate is 10 to 1,000 times as potent as nitrate (BPJ=300), many common foods contain substantial quantities of nitrate. Thus, single daily servings of many common foods are predicted to cause much more IUI than is predicted to result from two liters of drinking water at USEPA's proposed 1 ppb $DWEL_P$. Under the BPJ scenario, a single daily serving of milk correlates to 2*x* the 1 ppb $DWEL_P$; processed meats up to 100*x*; and vegetables up to 800*x*. The perchlorate-equivalent effect of consuming daily one serving of milk, one serving of 14 of 15 different processed meats, or one serving of 33 of 35 different vegetables would implicitly exceed that caused by two liters of drinking water at the $DWEL_P$. These results are illustrated in Figure 1a (milk and processed meats) and Figure 2a (vegetables). The thick gray lines in each figure at 10^0 represent IUI equivalent to that two liters of drinking water at the proposed 1 ppb $DWEL_P$. A single daily serving of cured ham, corned beef, broccoli, celery or lettuce would have about the same IUI effect as two liters of drinking water with 100 ppb perchlorate.

Even if it is assumed that perchlorate is 1,000 times as potent as nitrate, consumption of a single serving of 10 of 16 different processed meats and 30 of 35 different vegetables is predicted to cause more IUI than two liters of drinking water at the proposed 1 ppb $DWEL_P$. One serving of milk is predicted to have 40% of this IUI effect. A single daily serving of cured ham, corned beef, broccoli, celery or lettuce is predicted to cause about the same IUI effect as two liters of drinking water with 30 ppb perchlorate. This PER scenario probably under-predicts the relative magnitude of IUI caused by daily consumption of common foods containing nitrate.

These comparisons can be readily converted into the maximum number of servings that may be consumed per day and still avoid the level of IUI implied by USEPA's proposed 1 ppb $DWEL_P$. These comparisons are illustrated in Figure 1b (milk and processed meats) and Figure 2b (vegetables). Under the BPJ scenario, daily milk consumption must stay below 0.7 serving (6 oz; 170 g). Thirteen of 15 processed meats and 31 of 35 vegetables must stay below 0.5 serving. The exceptions are kimchi (6 servings), garlic (1.7 servings), mushroom and artichoke (0.6 serving each). Consumption of 25 vegetables must be limited to less than 0.1 serving, with 20 of these limited to less than 0.05 serving.

Using again the highest PER scenario (1,000), daily milk consumption must stay below 2.4 servings to yield predicted IUI below USEPA's proposed 1 ppb $DWEL_P$. Consumption of 10 of 15 processed meats must stay below one serving, and 8 of these 10 must stay below 0.5 serving. For 8 vegetables, consumption must be limited to less than 0.05 serving.

These figures show clearly why USEPA's 2002 draft perchlorate risk assessment radically overstates human health risks from perchlorate. Vegetable consumption is acknowledged as safe and healthful—even nitrate-rich green, leafy vegetables. Yet, if USEPA's 2002 perchlorate risk assessment is correct, then this comparison indicates that significant adverse thyroid effects should be occurring from nitrate in routine vegetable consumption, let alone nitrate in drinking water. However, adverse effects in humans from dietary nitrate have not been observed, even at much higher consumption rates than single daily servings. Thus, USEPA's proposed perchlorate RfD significantly under-predicts how much perchlorate exposure is "likely to be without an appreciable risk of deleterious effects during a lifetime." The origin of this inconsistency is USEPA's policy decision to deem IUI a "key event" potentially leading to changes in thyroid hormones and to derive the RfD as if such an event were itself adverse. This policy choice implicitly treats IUI as an adverse effect. Unless large and sustained over a long period, however, IUI is mundane and reversible, and arises at exposure levels orders of magnitude below the true adverse effect of interest. IUI cannot be *per se* adverse if it occurs routinely, reversibly and without ill effect at these levels at virtually every healthy meal.

This comparison also reveals that if IUI is implicitly deemed adverse and USEPA's 2002 draft $DWEL_P$ is necessary to protect against it, then USEPA's nitrate risk assessment cannot be accurate. For 10 of 15 processed meats and 30 of 35 vegetables, scientific uncertainty about the true value of PER is unimportant. Under any scientifically plausible PER scenario, single daily servings have IUI effects much greater than two liters of drinking water containing 1 ppb perchlorate. A comparison based on total dietary IUI would greatly intensify this discrepancy.

5.2 COMPARISONS BASED ON THE NITRATE MCL

As indicted above, USEPA's proposed 1 ppb $DWEL_P$ also can be compared with the existing nitrate MCL (10,000 ppb). Note these standards are different. The latter is a legally enforceable federal regulation that may not be exceeded and which incorporates other factors besides science, including cost, feasibility and affordability. The former is implied by USEPA's proposed perchlorate RfD, and is consistent with the interpreta-

tion USEPA has informally recommended and some State regulatory agencies have applied based on this recommendation. It does not take account of cost, feasibility or affordability. As we note in Section 8 below, these distinctions may not be important.

More importantly, $DWEL_P$ is intended to protect against IUI as a precursor of potential thyroid effects for which the risk assessment underlying the nitrate MCL did not take account (composite UF = 1). If the perchlorate risk assessment is valid and reliable, this analysis predicts IUI from dietary nitrate exposure hundreds or thousands of times greater than that predicted at the $DWEL_P$.

Figure 1a includes small vertical hash marks to the left of the band representing the perchlorate dose-equivalents from single daily servings of milk or processed meats expressed in ppb perchlorate in drinking water. These hash marks indicate the perchlorate dose-equivalents implied by PER = 10,000, the ratio of the nitrate MCL to the proposed perchlorate DWEL ($MCL_N/DWEL_P$). If USEPA or any State were to use $DWEL_P$ as a *de facto* drinking water standard, then this ratio would be analogous to an implicit PER. The hash marks tell us how much IUI is predicted to result from consuming single daily servings of common foods containing nitrate, in perchlorate dose-equivalents, under an implied PER of 10,000. The comparison is useful for discerning whether the two standards could possibly co-exist with any logical, albeit non-scientific, consistency. If single daily servings contain so much nitrate that predicted dietary IUI exceeds $DWEL_P$ even under the (non-scientific) assumption that perchlorate is 10,000 times as potent as perchlorate, it is reasonable to conclude that co-existence cannot be rationalized.

Figure 1a shows that under this extreme scenario, single daily servings of milk and 14 of 16 processed meats are predicted to cause IUI at levels below 1 ppb perchlorate dose-equivalent per day. However, there are two processed meats—corned beef and cured ham—for which single daily servings are predicted to exceed this level of IUI by about a factor of two. Figure 1b presents the same information in terms of the maximum number of servings that may be consumed per day and remain below the 1 ppb perchlorate dose-equivalent threshold. Approximately 20 servings of milk per day, or between two and 10 servings of eight different processed meats, are predicted to yield the same IUI as 1 ppb perchlorate in two liters of drinking water. Daily consumption of corned beef or cured ham, however, must stay below about 0.5 serving.

Analogous but more striking results are predicted for vegetables and are illustrated in Figures 2a and 2b. Figure 2a shows that, even under this extreme, non-scientific scenario, single daily servings of 18 of 35 vegetables are predicted to cause IUI equal to or greater than the amount predicted to result from 1 ppb perchlorate in two liters of drinking water. A single daily serving of rhubarb is predicted to cause about 10 times as much IUI; one serving of turnip greens about 20 times as much. Presenting the same information differently, Figure 2b shows that there are 13 vegetables for which between one and 10 daily servings are predicted to yield IUI exceeding this level. Single daily servings of 17 vegetables are predicted to exceed this amount. The vegetables in this latter group include many that are routinely consumed at levels well over one serving per day, such as salad greens (*e.g.*, endive, celery, lettuce and spinach), cruciferous vegetables (*e.g.*, cauliflower and broccoli), and other greens that are especially popular among members of certain subpopulations (*e.g.*, cabbage, kale/collard and turnip greens).

As indicated earlier, comparisons to the nitrate MCL are not scientific because there is no evidence suggesting that perchlorate is 1,000 times as potent as nitrate, much less 10,000 times as potent. Nevertheless, the comparison shows that MCL_N cannot logically stand if USEPA's 2002 draft perchlorate risk assessment is valid. For the nitrate risk assessment to also remain (barely) valid technically, IUI cannot below 10,000 ppb. But MCL_N does not provide much protection against IUI as a surrogate for thyroid effects. Dietary nitrate alone, much less nitrate from drinking water, is predicted to cause IUI at levels that USEPA's 2002 draft perchlorate risk assessment concludes are not protective. Consistency would require a substantial reduction in the nitrate NOAEL and MCL_N (both 10 mg/L). A reduction to 300 ppb would at least make the nitrate NOAEL and MCL consistent with USEPA's 2002 draft perchlorate risk assessment under the BPJ scenario. Reductions to 100 ppb or 30 ppb would be implied by precautionary PERs of 100 and 30, respectively.

6. Implied Risk Communication Messages

Based on this analysis, consumption of single daily servings of many common foods is predicted to result in much greater IUI than would be permitted by USEPA's proposed 1 ppb $DWEL_P$. If perchlorate exposure above this level is believed to be unsafe, then consumption of these foods also is unsafe—and by a much greater margin. However, if consumption of these foods is safe, then exposure to perchlorate levels hundreds or thousands of times greater than USEPA's proposed $DWEL_P$ also must be safe, and USEPA's perchlorate risk assessment is scientifically invalid and unreliable.

It is entirely reasonable to expect State regulatory agencies, the press and consumers at large to interpret USEPA's proposed 1 ppb $DWEL_P$ as the upper-bound level of exposure that is "safe," and to interpret exposures above this level as "unsafe" or perhaps "dangerous" [23-25]. USEPA does not actually state that such interpretations are correct, and for some chemicals it establishes enforceable health-based exposure levels well above the applicable RfD. Nevertheless, USEPA also does not generally expend effort countering these misunderstandings. Thus, it is instructive to examine the risk communication messages that are implied in USEPA's 2002 draft perchlorate risk assessment. This can be easily accomplished by converting results into the dichotomous categories "safe" and "unsafe," just as lay members of the public are likely to do.

Table 3 summarizes results for vegetables. Single-serving perchlorate dose equivalents (in ppb perchlorate in drinking water) from nitrate are reported to one significant digit for PER values ranging from 10 to 1,000, with intermediate categories that capture geometric midpoints. These values are formatted for three alternative risk messages: > 1 ppb, > 200 ppb, or > 500 ppb is "unsafe." Only eight of 175 (5%) vegetable-PER combinations that are double-underlined would be interpreted as "safe" under USEPA's proposed 1 ppb $DWEL_P$. Another 81 (46%) vegetable-PER combinations would be "safe" under a 200 ppb $DWEL_P$; these values are formatted in normal Roman font. Nineteen *italicized* vegetable-PER combinations (11%) would be interpreted as "unsafe" unless the $DWEL_P$ is increased to 500 ppb. For the 41 (23%) values in **boldface**, even a 500 ppb $DWEL_P$ is insufficient. Note that one serving of turnip greens is "unsafe" except under the PER scenario that predicts perchlorate is most potent relative

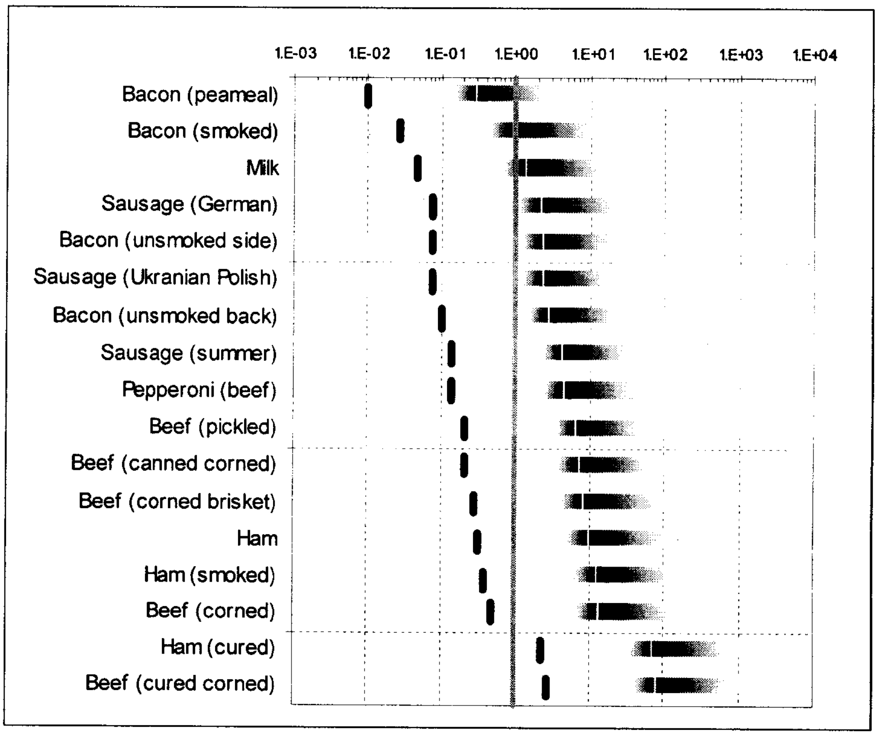

Fig 1a: Perchlorate dose-equivalents from single daily servings of milk or processed meats expressed in ppb perchlorate in drinking water.

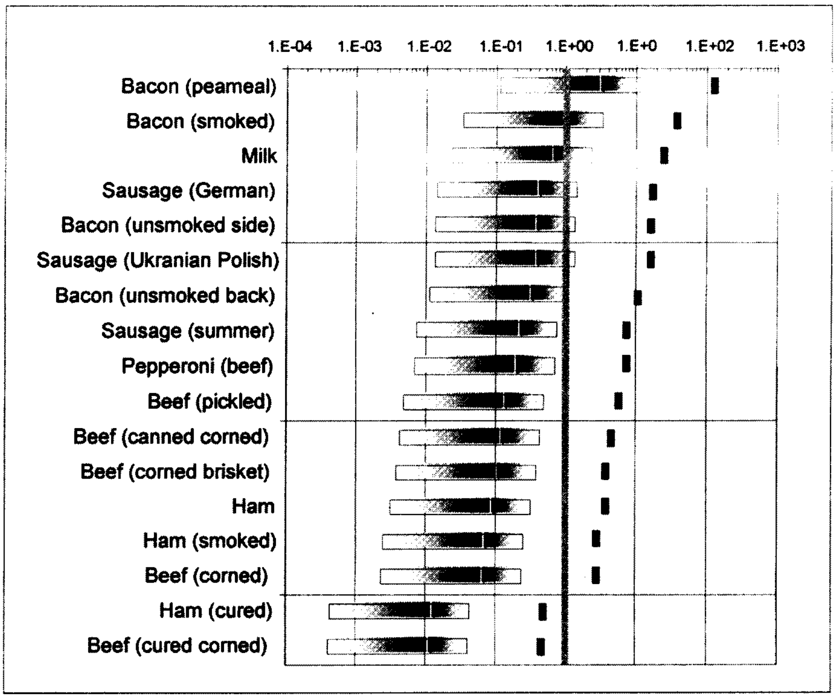

Fig. 1b: Predicted maximum servings of milk or processed meats permitted without exceeding 1 ppb perchlorate dose-equivalent in drinking water

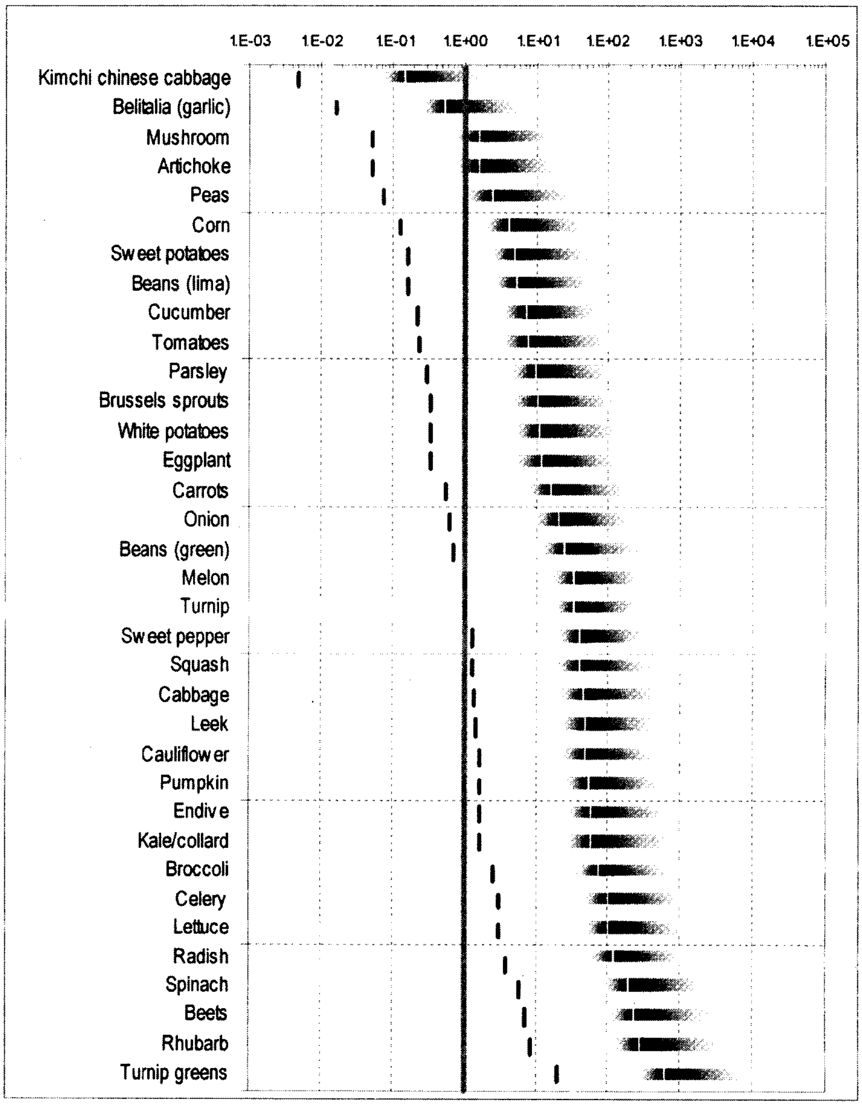

Fig 2a: Perchlorate dose-equivalents from single daily servings of vegetables expressed in ppb perchlorate in drinking water.

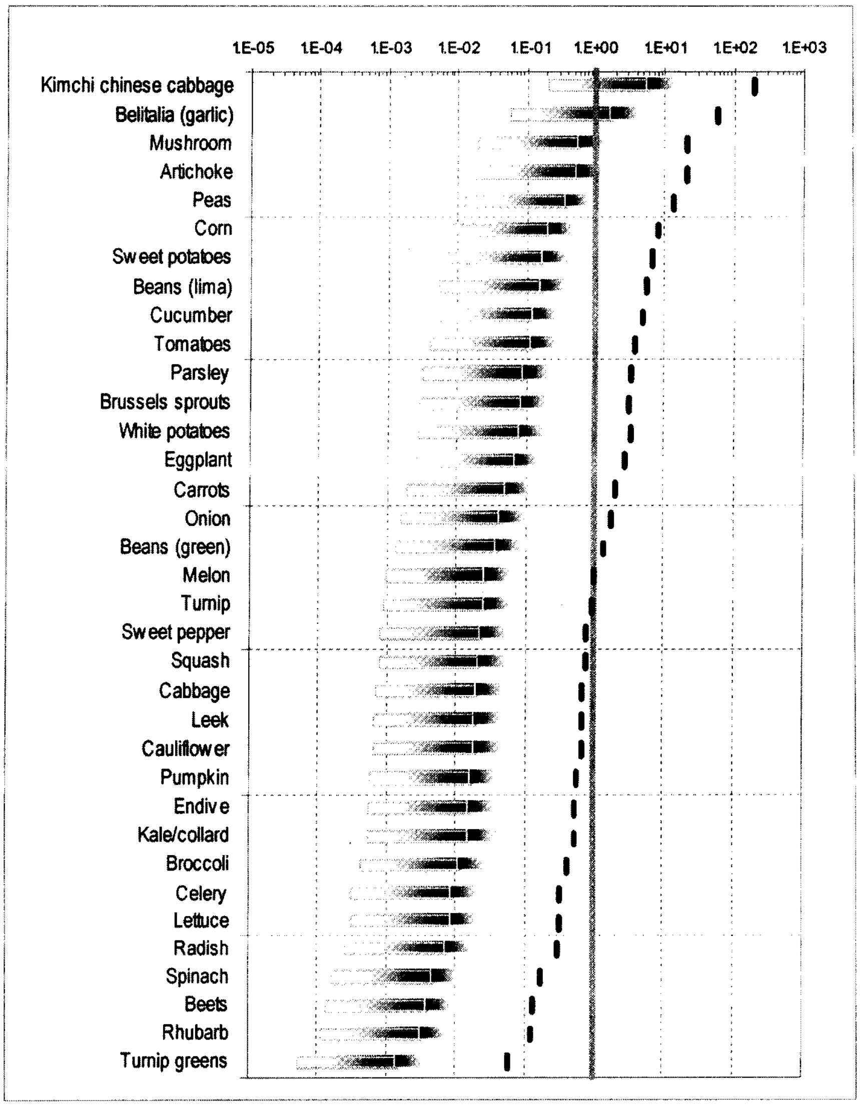

Fig 2b: Predicted maximum servings of vegetables permitted without exceeding 1 ppb perchlorate dose-equivalent in drinking water

TABLE 3: Approximate perchlorate-dose equivalents (in ppb perchlorate in drinking water) of single servings of vegetables and implied risk messages corresponding to alternative $DWEL_P$ of 1 ppb, 200 ppb, or 500 ppb. Dose equivalents are based on a PER range of 10 to 1,000, with a BPJ value of 300.

PER Value	10	30	100	300	1000
Kimchi	6.	2.	1.	.2	.1
Belitalia (garlic)	20.	7.	2.	.7	.2
Mushroom	60.	20.	6.	2.	.6
Artichoke	70.	20.	7.	2.	.7
Peas	100.	30.	10.	3.	1.
Corn	200.	60.	20.	6.	2.
Sweet potatoes	200.	70.	20.	7.	2.
Beans (lima)	200.	80.	20.	8.	2.
Cucumber	*300.*	100.	30.	10.	3.
Tomatoes	*300.*	100.	30.	10.	3.
Parsley	*400.*	100.	40.	10.	4.
Brussels sprouts	*400.*	100.	40.	10.	4.
White potatoes	*500.*	200.	50.	20.	5.
Eggplant	*500.*	200.	50.	20.	5.
Carrots	**700.**	200.	70.	20.	7.
Onion	**800.**	*300.*	80.	30.	8.
Beans (green)	**1,000.**	*300.*	100.	30.	10.
Melon	**1,000.**	*500.*	100.	50.	10.
Turnip	**1,000.**	*500.*	100.	50.	10.
Sweet pepper	**2,000.**	*500.*	200.	50.	20.
Squash	**2,000.**	**600.**	200.	60.	20.
Cabbage	**2,000.**	**600.**	200.	60.	20.
Leek	**2,000.**	**700.**	200.	70.	20.
Cauliflower	**2,000.**	**700.**	200.	70.	20.
Pumpkin	**2,000.**	**800.**	200.	80.	20.
Endive	**2,000.**	**800.**	200.	80.	20.
Kale/collard	**2,000.**	**800.**	200.	80.	20.
Broccoli	**3,000.**	**1,000.**	*300.*	100.	30.
Celery	**4,000.**	**1,000.**	*400.*	100.	40.
Lettuce	**4,000.**	**1,000.**	*400.*	100.	40.
Radish	**5,000.**	**2,000.**	*500.*	200.	50.
Spinach	**8,000.**	**3,000.**	**800.**	*300.*	80.
Beets	**10,000.**	**3,000.**	**1,000.**	*300.*	100.
Rhubarb	**10,000.**	**4,000.**	**1,000.**	*400.*	100.
Turnip greens	**30,000.**	**8,000.**	**3,000.**	**800.**	*300.*

[double underline: > 1 ppb is "unsafe"] [*italics*: > 500 ppb is "unsafe"]
[Roman: > 200 ppb is "unsafe"] [**bold**: >> 500 ppb is "unsafe"]

to nitrate (PER = 1,000). However, this PER likely over-predicts the potency of perchlorate relative to nitrate, because it exceeds the highest ratio reported in any study. Even if a PER of 1,000 is interpreted as the best value for this comparison, perchlorate dose-equivalents corresponding to single daily servings of 30 of 35 commonly consumed vegetables still exceed the proposed $DWEL_P$ of 1 ppb.

Ninety-five percent of the single-serving vegetable-PER combinations would be widely interpreted as "unsafe" under USEPA's proposed 1 ppb $DWEL_P$. Half would be "unsafe" at an alternative $DWEL_P$ of 200 ppb. Both risk messages defy common sense. A $DWEL_P$ of about 800 ppb is necessary just to ensure that a single daily serving of turnip greens is not misinterpreted as "unsafe." Indeed, any total diet that is not strictly carnivorous would be "unsafe" even at this value.

7. Acknowledgements

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8. References

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COMPARISON OF RISKS FROM USE OF TRADITIONAL AND RECYCLED ROAD CONSTRUCTION MATERIALS: ACCOUNTING FOR VARIABILITY IN CONTAMINANT RELEASE ESTIMATES

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Abstract

Recycled materials, such as recovered materials from the transportation sector or secondary or by-product materials from the industrial, municipal, or mining sectors can be used as substitutes for natural materials in the construction of highway infrastructure. Trace metals in these recycled materials may leach out and contaminate the groundwater and soil posing a long-term environmental problem. Environmental risk assessments are necessary to evaluate which recycled material applications are acceptable. The first step for determining the environmental risk of using recycled materials is to characterize the source term. Estimates of contaminant release fluxes can then be used in a comparative risk assessment. This paper will give an example of a comparative, probabilistic approach for exposure assessment. Existing deterministic models for estimating contaminant release will be presented and incorporation of variability in these models will be discussed.

1. Introduction

In the U.S. alone, there are six million kilometers of roads [1]. Large volumes of materials and thus significant quarrying are required for construction and maintenance of these roads. A more sustainable alternative to using traditional materials is to use recycled materials in roads. Further utilization of otherwise waste materials redirects the path of these materials from landfills to their beneficial use as embankments or as surface, base, or subbase layers in roads.

For secondary materials to be recycled in roads, they need to have good engineering and environmental properties. Some of the more common recycled materials are recycled asphalt pavement, reclaimed concrete pavement, scrap tires, and coal combustion and steel production by-products. In the U.S., use of these recycled materials can meet the demand for a significant portion of the large volumes of construction materials needed for road building and maintenance every year [2]. Yet, the U.S. is not utilizing its full potential to recycle, especially when compared to Europe [3]. For example, municipal solid waste incinerator (MSWI) bottom ash is a

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valuable commodity and completely recycled in Europe for its use as road construction material; but in the U.S., MSWI ash is landfilled.

A major barrier to recycling in the roadway environment in the U.S. is the lack of information on potential ecological and human health risks [4]. The regulators are unwilling to take any possible risks from using recycled materials even though they may already be taking them when they use traditional materials. The long-term risk of concern to the state departments of environmental protection is the potential leaching of contaminants such as trace metals and organics. These contaminants may be found both in traditional and recycled materials, yet the fear of creating linear landfills and contaminating the groundwater is associated mainly with recycled materials.

To improve recycling in the U.S., the risk from traditional and recycled materials need to be compared. The comparison is especially needed for relatively new recycled materials (e.g. aluminum dross, waste vinyl plastics, construction and debris fractions, dredged sediments) to determine if it is worthwhile to invest time and research in developing high quality engineering products out of these less traditional candidates. A comparative risk assessment study of traditional and newly proposed recycled materials is most dependent on the relative magnitudes of the contaminant release as it is mainly the exposure step of a risk assessment where differences between traditional and recycled materials will be observed. The objective of this paper is to discuss (1) available methods for estimating the source term and (2) broaden current approaches by suggesting a probabilistic outlook to incorporate variable pavement designs in contaminant release estimates. A comparative example is given to illustrate the probabilistic approach.

2. Extrapolating from Lab Experiments: 1D Diffusion model

Contaminant release depends on contaminant solubility, diffusion, and advection. As contaminants solubilize, they diffuse within the particle pore space and across the aqueous boundary layer that surrounds the particle. If the hydraulic regime is governed by "fast" fluxes, advection will quickly remove the released contaminant from the source, thereby leaving the solution unsaturated and allowing more release. If advection is slow, such as in a slow percolation system through unbounded materials, then the solubility may govern the maximum concentration of release. In granular materials (base course, embankments), the release is more often controlled by solubility. In monolithic systems (asphalt concrete, Portland cement concrete), the rate limiting step in release of contaminants is more typically diffusion.

One approach that has been commonly used in Europe and may be widely adopted in regulations in the U.S. as well [5], is to use the one dimensional diffusion equation to estimate contaminant release from monoliths [6]:

$$M = \frac{2C_0}{height}\left(\frac{D_{obs}time}{\pi}\right)^{0.5}$$

Where,

M = contaminant mass released, mg contaminant/kg material,
C_0 = initial available concentration (mg contaminant/kg material),
Height = height of application
D_{obs} = observed diffusivity (m^2/s)
Time = lifetime of a pavement (seconds)

The height of application and the lifetime of the pavement are design-specific. The initial available concentration and the observed diffusivity are determined by standardized tank leaching and availability laboratory tests [7-8]. To account for unsaturated periods where diffusion may be less or nonexistent, the cumulative release can be normalized to the period of time that the material was wet [6].

In the waste research realm, little effort has been made to systematically incorporate variability and uncertainty in contaminant release estimates. The uncertainty arises partly from the simplicity of the approach. One dimensional contaminant release estimate may be overly conservative when diffusion occurs towards all wet surfaces and the release may be to the top of the application in addition to being towards the groundwater table. Similarly, the initial available concentration is only an operational definition. The standardized test may not approximate the true value of the available concentration in the field. In addition to model errors, significant uncertainty also exists in accurate and precise determination of the initial available concentration and the observed diffusivity. Within lab and between lab variability has been documented [9,10], yet rarely used in the literature.

There is significant variability in recycled material properties and pavement designs. Accounting for this variability has so far meant calculating different scenarios. A more informative and systematic method than documenting individual scenarios is to treat the parameters of the diffusion equation probabilistically. Accounting for variability in this way is computationally simple because only four parameters and a single equation are required.

As an illustrative example, data from de Groot et al. [11] was used to compare leaching of arsenic from Portland cement concrete and asphalt concrete, both of which were made with traditional and recycled materials. Arsenic was selected for this example because (1) it is a common contaminant in asphalt concrete and Portland cement concrete and (2) information on potential sources of groundwater arsenic contamination is meaningful when the MCL of arsenic in drinking water has recently (February 2002) been reduced to 10- ppb.

The lifetime of the pavement was assumed to be a normal distribution with a mean of 15 years and a standard deviation of five years. This distribution was selected to account for variability in the climate (i.e. different geographical locations), volume of traffic, and the pavement design. The height of the surface layer in pavements may vary from 10cm to 40 cm in the presence of overlays. Thus, a uniform distribution from 0.1m to 0.4m was selected to represent the variability in height of the application in existing and future designs. Observed diffusivity was represented by a lognormal distribution. Measurement of observed diffusivity values are also often reported in log units. De Groot et al. [11] reported observed diffusivity and initial available concentrations of Portland cement concrete and asphalt concrete with and without

addition of different types of recycled materials (coal neutral and basic fly ash and MSWI bottom ash). The distributions selected for observed diffusivity and initial available concentration based on the values reported in de Groot et al. [11] are shown in Table 1. The observed diffusivity and initial available concentration of arsenic in asphalt concrete and Portland cement concrete without recycled materials was not modeled stochastically because there was not available data to define a probability density function. For materials containing recycled materials, a uniform distribution was selected for initial available concentration because there was no basis to select a different distribution. The variability in parameters was propagated across the one-dimensional diffusion equation using a one-dimensional Monte Carlo experiment with Latin hypercube sampling.

TABLE 1: Observed diffusivity and initial available concentrations.*

	Portland Cement Concrete		Asphalt Concrete	
	w/ recycled materials	w/o recycled materials	w/ recycled materials	w/o recycled materials
Observed diffusivity (m^2/s)	Lognormal ($3.16\ 10^{-10}$, $2.76\ 10^{-10}$)	$8.51\ 10^{-12}$	Lognormal ($8.42\ 10^{-13.5}$, $5.4\ 10^{-13}$)	$6.31\ 10^{-12}$
Initial available concentration (mg/kg)	Uniform (0.03, 0.05)	0.012	Uniform (0.04, 0.09)	0.03

* The values in parentheses represent mean and standard deviation for lognormal distribution, and upper and lower values for uniform distribution.

The results of the simulations are shown in Figures 1 and 2 in the form of cumulative variability where the arsenic release at 100% variability represents the worst case scenario among all parameters. If the model is assumed to represent reality, one can say that 90% of all existing pavement designs would release no more than 0.011mg/kg arsenic for asphalt concrete pavements and no more than 0.125mg/kg arsenic for Portland cement concrete pavements. This approach can also be used to determine what percent of designs may exceed a given contaminant release limit. In this example, there was higher release in the presence of recycled materials from both Portland cement concrete and asphalt concrete.

A meaningful way to interpret the release information is to compare the contaminant release to background soil concentrations. The range of arsenic background soil concentrations in a few states is shown in Figure 3. Depending on the location of the road, the release may be a very small burden to the soil. For example, in Colorado, the total release from the road is expected to be 15 times less than the lower value of arsenic found in the soil. (This analysis assumes that all arsenic released is retained in the upper soil horizon, with a depth equal to the depth of recycled material application above the soil.)

The simple, probabilistic approach presented here and illustrated with an example provides a powerful method to compare the potential risks from use of traditional and recycled materials in roads. Since use of the one dimensional diffusion equation approach is simple by itself, efforts may focus on incorporating variability and

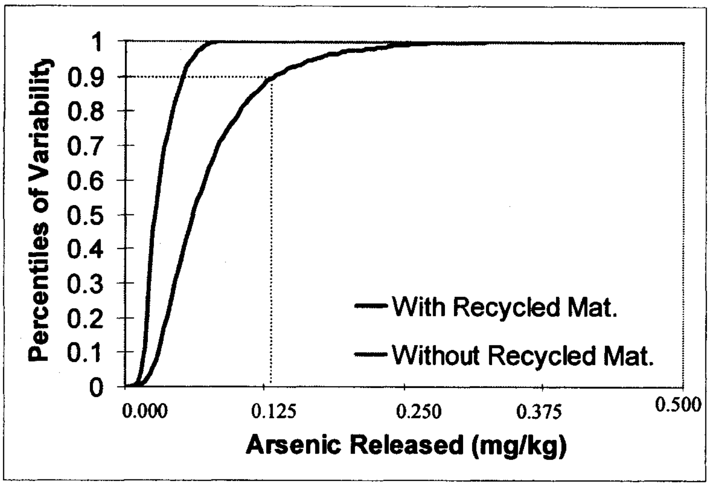

FIGURE 1 Arsenic release from Portland cement concrete for different designs

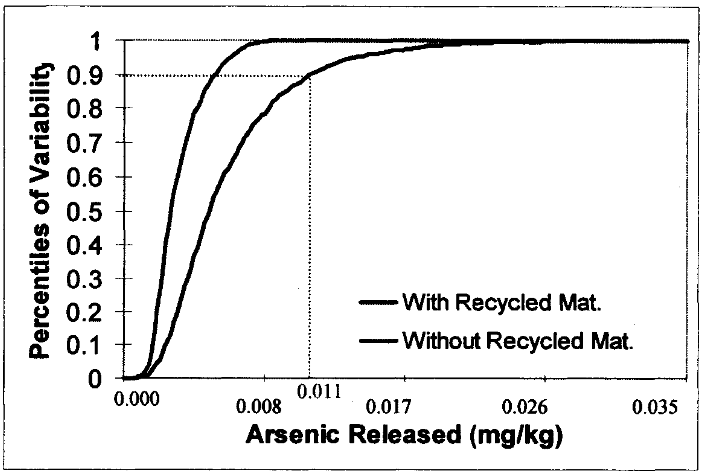

FIGURE 2 Arsenic release from asphalt concrete for different designs

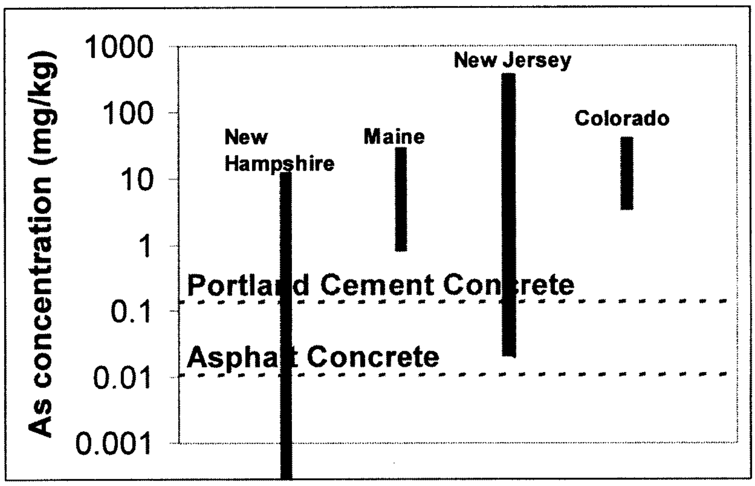

FIGURE 3: Background arsenic soil concentrations [12]

uncertainty in the parameters. For general comparisons of different designs or recycled versus traditional materials, better and more informative estimates of contaminant release may be obtained by using the probabilistic approach as opposed to analyzing selected scenarios.

3. Discussion

To be able to use the one dimensional diffusion equation, good understanding of hydraulic regimes in the pavement is necessary. The material in the pavement may remain dry even during a precipitation event if the hydraulic conductivity of the material itself or the material around it is sufficiently low. Unfortunately, the hydraulic regimes in pavements are not well characterized although there is ongoing research in this area [13]. To better understand water movement in roads and also to estimate contaminant release fluxes, unsaturated contaminant transport models can be used (e.g. HYDRUS2D). Use of these models provides better representation of temporal and spatial scales. For example, different layers in a pavement, the effect of paved and unpaved shoulders, cracking, water collection in drainage pipes installed in the pavement can all be represented in detail using a finite element or finite difference code. Contaminant transport models can also simulate solubility/availability limited rates of contaminant release for periods or pavement sections where contaminant release is not limited by diffusive mass transfer.

Unsaturated contaminant transport modeling is more informative and realistic, however, it requires significantly more effort in both data collection and model evaluation steps. In addition to initial available concentration and observed diffusivity,

hydraulic properties of the pavement materials and detailed knowledge on cracks and geometry are required but often this information is not available. Because the model itself is complex, it is not as easy to account for uncertainty and variability as it is using the one dimensional diffusion equation.

4. Conclusion

The use of recycled materials is perceived as risky even though similar risks may be present from the use of traditional construction materials. Lack of comparative information on contaminant release is the major inhibitor to utilizing secondary materials in roads. The biased and unfavorable risk perception on recycled materials may change if long-term contaminant release from traditional and recycled materials can be realistically predicted. This paper demonstrated empowering a simplified approach by easily incorporating variability in the model parameters. Use of Monte Carlo simulations to propagate variability between pavement designs is new in the waste research realm. On the other extreme of modeling is an unsaturated contaminant transport model that would yield more detailed information on spatial and temporal scales but would require much greater effort to implement and account for uncertainty and variability. Future research should emphasize accounting for variability and uncertainty in both the simple and the more rigorous approaches for estimating contaminant release from road construction materials. Only in this way can the probabilistic approaches that have almost become standard in other components of risk assessment may make their way to the source term estimates.

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ENVIRONMENTAL RISK ASSESSMENT OF PESTICIDES IN NEPAL AND HINDUKUSH-HIMALAYAN REGION

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Abstract

An environmental risk assessment, uncommon since focusing only on the pollution risk of soil and groundwater through pesticides, for a site in the Mid-Hills of Nepal will be introduced and discussed.

It will be shown, that model fits which have been developed for the three compounds Metalaxyl, Dimethoate and Fenvalerate, show satisfying results for the further application on risk assessment for the first two, as they account for the general trend of the transport mechanism. For the latter one, the applied model failed due to unidentified transport and degradation processes. Furthermore, the assessment profiles show phenomena in the data which cannot be explained with the applied model, i.e. higher concentrations of pesticides in greater depths than in medium depths. Field studies conducted as combined tracing experiments with the respective pesticides and Vitasin Blue FCF 90 and Deuterium (as a conservative tracer) are used to investigate these phenomena. Results show a leading role of preferential flow paths and free soil water in the transport of pesticides into the deeper soil and into the groundwater system under the given agricultural system (ponding and furrow irrigation), climate and soil type. These additional field studies were necessary in order to reach a sound environmental risk assessment.

1. Introduction

Misuse of pesticides is an increasing problem in developing countries, also in the Hindu Kush-Himalayan (HKH) Region and in Nepal. To investigate risks of pesticides in a subtropical environment an interdisciplinary collaborative research project was set up in 1999 ending in 2002 (*"Environmental risks of pesticides and sustainable development of integrated pesticide management for mountain areas of developing countries considering socio-economic conditions and taking Middle Mountains, Central Nepal as an example"*). It was funded by the Volkswagen Stiftung Foundation, Germany. Besides pure research partners, governmental institutions and non-governmental organisations (NGO's) participated in the project in order to account for knowledge

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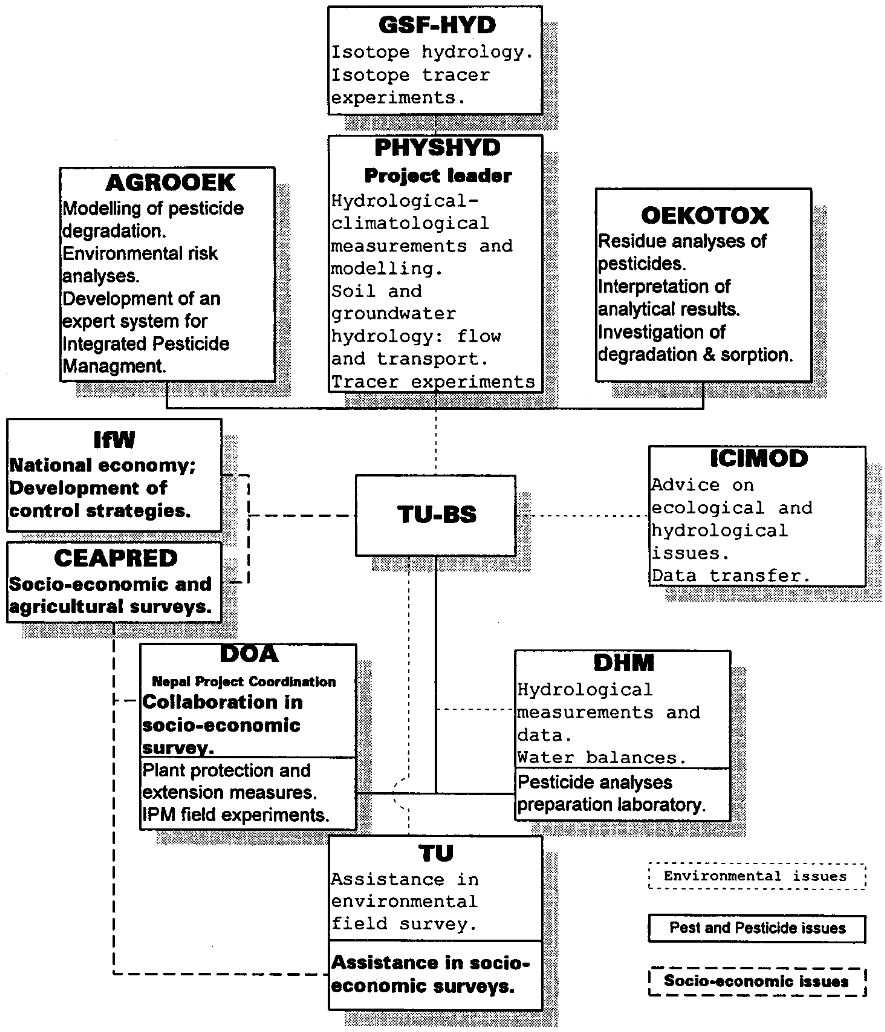


Fig. 1. Organisation of the project with main activities of collaborating partner institutions on environmental, pesticide and socio-economic issues; German Partners-: TU-BS: Technical University of Braunschweig, AGROOEK: Institute of Geoecology, Department of Agroecology, PHYSHYD: Institute Geoecology, Department of Hydrology and Landscape Ecology, OEKOTOX: Institute of Ecological Chemistry and Waste Analysis, GSF-HYD, GSF-Research Centre for Environment and Health Munich, Institute of Hydrology, IfW: Kiel Institute of World Economic., Nepalese Partners-: ICIMOD :International Centre for Integrated Mountain Development, DHM: HMG Department of Hydrology and Meteorology, DOA: HMG Department of Agriculture, CEAPRED: Center for Environmental and Agricultural Policy Research, Extension & Development, TU: Tribhuvan University–Central Geography Department

transfer and extension of results to farmers. The project's organisational structure is shown in Figure 1 highlighting on all participating partners and their major tasks. The focal research topics were: (1) *Environmental Issues*, concentrating on possible contamination and contamination paths in the environment (soils and water resources) by pesticides. (2) *Pest and Pesticide Issues* concentrating on pest identification and pest control through an Integrated Pest Management (IPM) and the analysis of pesticides including degradation and sorption experiments. (3) *Socio-Economic Issues,* concentrating on the socio-economic background of the farmers and the value of pesticides and pesticide use for the national economy. The project did not aim on risk analysis from the classical point of view as it did neither account for health risks of farmers through application practices nor for consumers' risks through consumption of field fruits or drinking water.

The project was finalized with an international workshop on environmental risk assessment in Kathmandu (Nepal) in order to present the project's results to scientists and Nepalese decision makers and to exchange knowledge with scientists of the region [4].

2. Geographical Background

The studies sites were situated at about 50 km east of Kathmandu along the Arniko highway in Jhikhu-Khola catchment of the Kavhre district. The good infrastructure of the area allows cash crop production for the nearby markets Kathmandu and Banepa. The climatic conditions, which are summarized in Figure 2, favour agriculture throughout the year when crops are irrigated. The terraced, irrigated lands, so called *khet* lands, give three harvests a year (e.g. potato, tomato or maize followed by rice) while rain-fed agricultural areas (*bari* lands) produce two harvests only. Pesticide application intensities and frequencies are higher under *khet* than under *bari* conditions. The soil substrates are loamy, with 45% sand, 37% silt and 18% clay in the first 20 cm. Organic carbon amounts to about 1%. With depth soil composition varies showing a sand fraction between 40-58%, silt between 30-37% and clay between 12-22%, in all cases though sand always dominating the substrate.

Irrigation takes place using the ponding method for field preparation works and the cropping of rice and using furrow irrigation for potatoes and tomatoes.

A simplified hydrological model as valid for the pesticide transport through the environment under *khet* conditions is shown in Figure 3. Highlighted are those compartments which are of special interest to the risk assessment for the environment, and which were subject to thorough scientific studies.

3. Data and Risk Modelling

In order to develop a risk analysis model that is based on reactive transport modelling for pesticides, that accounts for their degradation and sorption kinetics, and that could forecast risks of pesticide application under the given agricultural and climatic

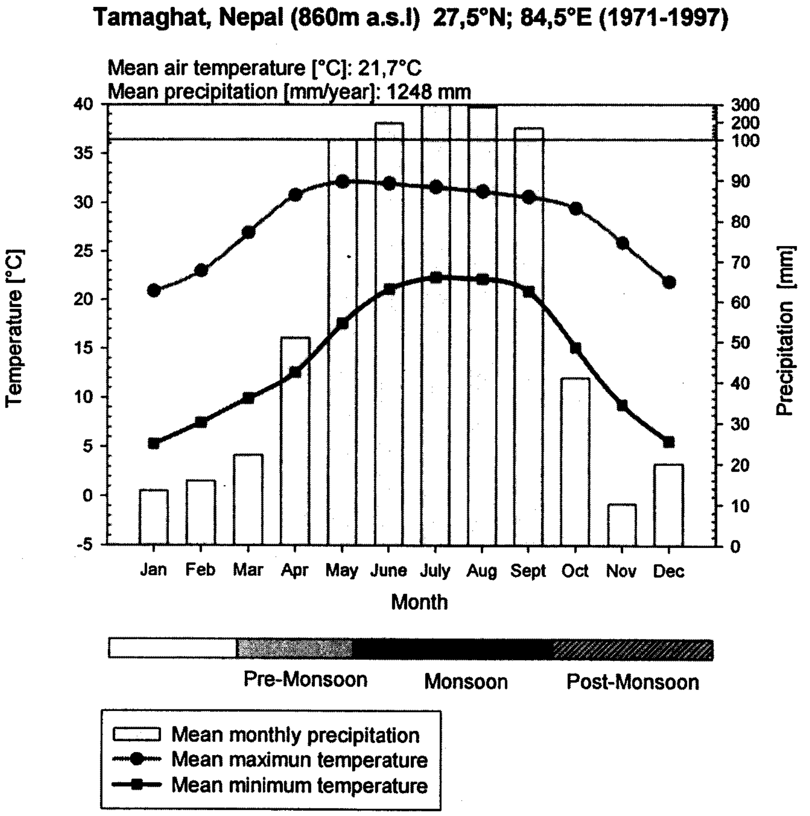
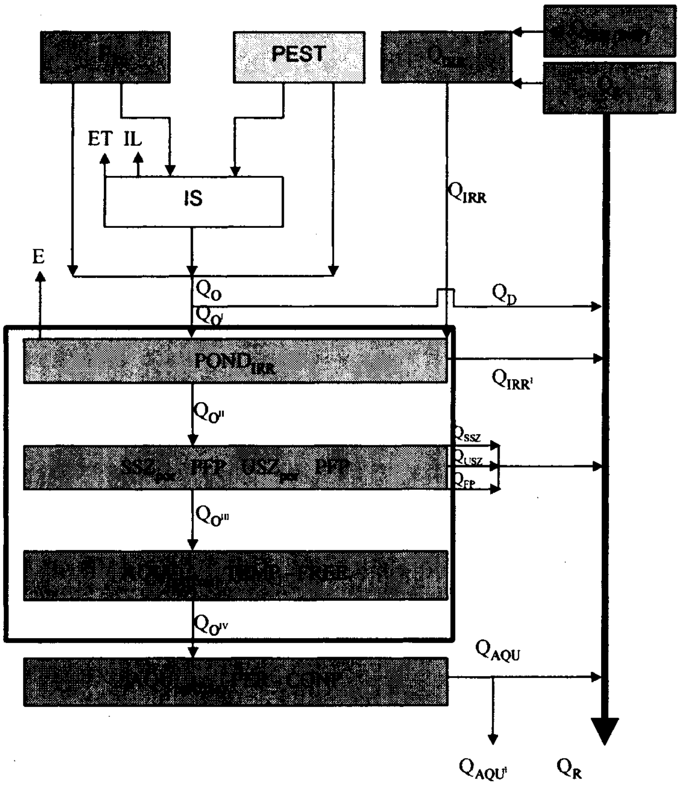


Fig. 2. Climatic conditions at the study sites, Jhiku-Kola Catchment, Nepal

conditions, a number of experiments on different scales needed to be carried out, in order to define required kinetics and water movements.

Under laboratory conditions, firstly the sorption and degradation kinetics of the selected pesticides Malathion, Dimethoate, Fenvalerate and Metalaxyl were determined in controlled batch and column experiments. As a follow up, investigations on one-dimensional transport of water and pesticides, using Bromide as a control media tracer, and investigations on pesticide degradation were carried out under field conditions on plot scale. Field data on soil moistures and climatic conditions was collected over a period of two and a half years in the project catchments, besides the data taken at plot scale.



$AQU_{por(frac)}$	Aquifer, porous (fractured)	Q_D	Surface Runoff
E	Evaporation	Q_{FP}	Flux from Preferential Flow Path
ET	Evapotranspiration	Q_{IRR}	Irrigation Water
IL	Interception Loss	$Q_{IRR(IMP)}$	Irrigation Water Channel, imported
IS	Interception Storage	Q_O	Infiltration\Seepage Flux
P	Precipitation	Q_R	River Water
PEST	Pesticide Application	Q_{SSZ}	Flux from Saturated Soil Zone
PFP	Preferential Flow Paths	Q_{USZ}	Flux from Unsaturated Soil Zone
$POND_{IRR}$	Irrigation Water, ponded	SSZ_{por}	Saturated Soil Zone
Q_{AQU}	Groundwater Flow	USZ_{por}	Unsaturated Soil Zone

Fig. 3. Simplified conceptual hydrological model

The risk analysis was based on the one-site kinetic sorption model for the description of degradation and sorption developed by Richter 1996 [1]:

$$\frac{d}{dt}(\theta c) = -\alpha_S \varrho (K_D c - S) - \theta k c$$

$$\frac{d}{dt}(\varrho S) = \alpha_S \varrho (K_D c - S)$$

c: solute concentration
S: sorbed concentration
α: sorption rate constant
ρ: soil bulk density
θ: volumetric water content
KD: sorption equilibrium constant
k(T, θ): dependent degradation rate

To account for transient water movement the Convection-Dispersion Equation was coupled with the modified Richards equation for the dual-porosity case (two capillary domain model) with convection and dispersion in the macro-pores and diffusion to the matrix [1].

To take care for the above expressed dependence of sorption and degradation kinetics on temperature and soil moisture (Equation 1), the pesticide degradation was tested in the laboratory [8]. Results for two of the compounds are shown in Figure 4. Temperature changes were set such that during the first 5 days the soil temperature was 20 °C, up to the 10^{th} day the soil temperature was 10 °C and until the 15^{th} day the soil temperature was 30 °C. The experiment was set for soil moistures of 20, 40 and >50 Vol.%. Important to note are in case of Dimethoate the significant response of degradation to temperature and humidity changes and the apparent decline of degradation with increasing soil moisture. Latter finding was unexpected since literature usually indicates the opposite due to rising microbial activity with soil moisture [3]. Fenvalerate shows no dependence of degradation on soil moisture but on soil temperature only.

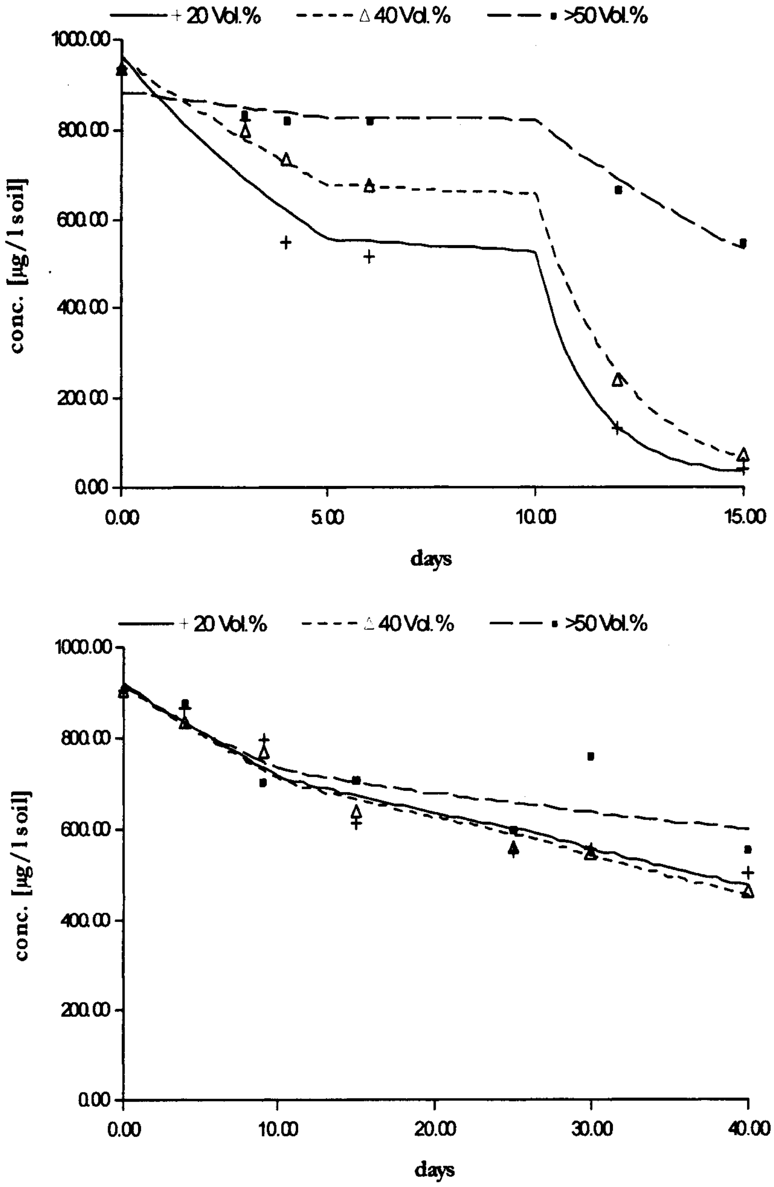


Fig. 4.: Data and model fit of Dimethoate (above) and Fenvalerate (below) with varying temperatures and different soil water contents; temperature change: 20°C/10°C/30°C; total R^2 = 0.97812 for Dimethoate and total R^2 = 0.9051 for Fenvalerate (From [1])

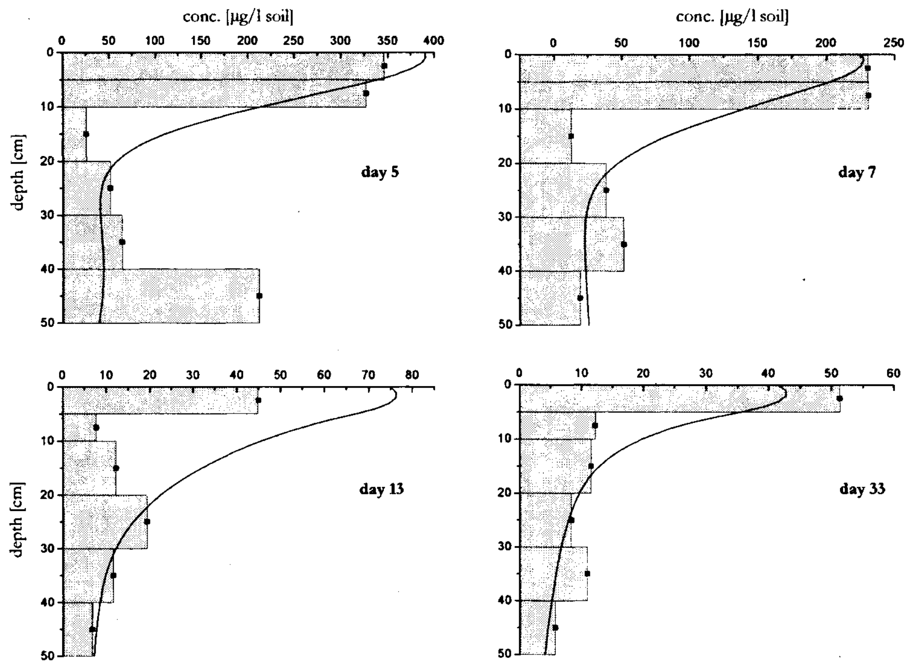


Fig. 5. Results of pesticide degradation and sorption in the field: Dimethoate profiles and model fits, field trial (type A) September 2000 (From [1])

Modelling was furthermore based on field experiments with pesticides and Sodium Bromide (NaBr) being used as a tracer (field experiments type A). These field experiments were conducted by pesticide and solved NaBr application on the bare ground of test plots of 60 m^2 size. The plots were thereafter ponded with irrigation water, methodically following rice cultivation praxis. Sampling was done using a N_{min}-drill with 5 sampling points per field in 0-5, 5-10, 10-20, 30-40 and 40-50 cm depth. The samples were mixed according to depths before analysis. The results for the Dimethoate sampling profiles at day 3, 7, 13 and 33 with their respective model fits are shown Figure 5. The model fit shows satisfying results for a further application on risk assessment, as it accounts for the general trend of the transport mechanism but the profiles also show phenomena in the data (e.g. low concentrations between 10-20 cm and high concentrations at 40-50 cm on day 5) which cannot be explained with the applied model.

Here shall be furthermore noted that model development was successful for the substances Dimethoate and Metalaxyl but failed for Fenvalerate.

4. Environmental Risk Assessment

For environmental risk analysis different scenario computations were carried out. The scenario selection was based on detailed information collected during a socio-economic survey carried out by CEAPRED being one of the project partners [2]. Table 1 lists the doses in terms of active ingredient per square meter and application intervals in days for three different scenarios.

The worst case scenario was based on the highest recorded dose for all crops exceeded by +1.96 associated standard deviations and the highest recorded crop specific application interval, while the best case scenario was based on the respective lowest values. The results based on the worst and best case scenarios as valid for Dimethoate are shown in Figure 6. For Dimethoate the approximate outflow per square meter and year would be 132 μg (best case) and 17929 μg (worst case). For Metalaxyl 32 $\mu g\ m^{-2}a^{-1}$ and 4340 $\mu g\ m^{-2}a^{-1}$ respectively. The vertical dotted lines indicate quantification limits, showing that in the best case scenario concentrations in the soil stay well below quantification possibilities while in the worst case scenario considerable concentrations are found throughout the soil column.

The overall results, based on a standard deviation, give a probability of 2.5% for the best and the worst case, respectively. These still needed to be corrected by the probabilities of the socio-economic survey, ending in probabilities for the worst case scenario of 0.416% [1]. As a general risk assessment for the watershed was concluded based on the modelling work, that no acute risk of residue formation or groundwater or open water contamination exists under the present situation, but that there are chances of a long term residue formation (>1%), especially with synthetic pyrethroids. [1]

TABLE 1: Dosage and application intervals of the four selected pesticides for the mean, worst and best case scenarios; application intervals for potato/tomato/rice cropping period (From [1])

	mean		worst case		best case	
	dose [μg a.i./m²]	application interval [d]	dose [μg a.i./m²]	application interval [d]	dose [μg a.i./m²]	application interval [d]
Malathion	86000	--/--/23	144996	--/--/18	27004	--/--/36
Dimethoate	25120	10/10/23	59408	7/7/23	8908	14/14/36
Fenvalerate	17000	10/10/23	34720	7/7/18	15464	14/14/36
Metalaxyl	7405	9/9/--	12848	7/7/--	2442	18/18/--

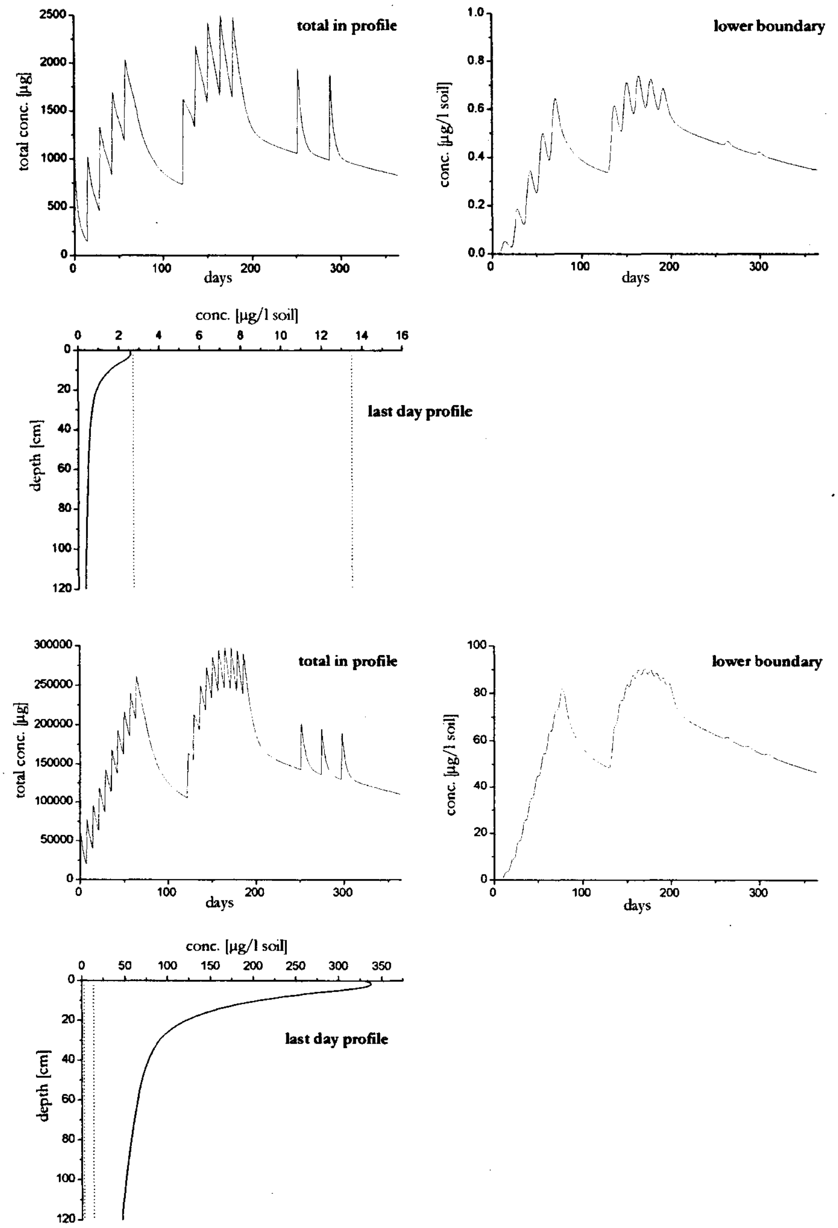


Fig. 6. Results of the best case scenario calculation (above) and worst case scenario calculation (below) for Dimethoate; reference area: 1 m², dotted lines in last day profile indicate quantification limits (From [1])

5. Detailed Transport studies

The field experiments on pesticide transport conducted up to a depth of 50 cm and based on soil sampling as described in section 3 showed, in some cases, higher concentrations of pesticides in greater depths than at medium depth (see Figure 5), resulting in model fit problems. Furthermore, modelling for the compound Fenvalerate failed completely due to unidentified transport mechanism. In order to investigate contamination risk and to back the modelling, studies on transport pattern into greater depths were conducted.

5.1 COMBINED PESTICIDE-VITASIN BLUE FCF TRACING EXPERIMENT

One experiment (B) was performed on four plots, each of the size of 2 m^2, where pesticides were applied on the bare soil, with concentrations for Dimethoate of 1200 g/ha, Fenvalerate 400g/ha, and Metalaxyl 400 g/ha. Thereafter a Vitasin Blue FCF 90 solution was applied (6250 kg/ha) and the plots were ponded. Conditions were as under rice cultivation. After 3, 10, and 21 days the plots were opened vertically such that sections were cut in the centre of the plot [5]. Results of the section of test plot number one (3 days after application) are found in Figure 7. It accounts only for the first 110 cm, but the section had been cut up to 250 cm reaching the uplifted groundwater table. It can be clearly seen, that water fluxes (dark, as indicated by the Vitasin Blue) have reached great depths. It is also indicated that blue sections reached down to 240 cm below surface; the uplifted groundwater table below the 240 cm was also reached by the Vitasin Blue. Sampling was done at spots indicating water fluxes, as well as on spots without any water flux indication (Samples 1-12).

The results show, independently from the substance-specific water solubility, a transport of the respective pesticides into the deeper soil [7]. This indicates clearly a risk of fast transportation of the respective pesticides Metalaxyl and Dimethoate, but also Fenvalerate into great depths within a limited period of time (<3 days) reaching down to the uplifted groundwater table, resulting in a potential pollution risk. Here, the formation of contaminant pools in greater depths (Figure 6) also adds. Unfortunately, a quantification of pesticides reaching the groundwater or being absorbed within the soil column could not be done though a linear relation between Vitasin Blue application and pesticides could be shown for the three-days experiment [5]. Since the quantification failed, no appropriate risk analysis was carried out.

Following the sampling procedure of experiment type A, samples (samples a to d) have been taken additionally with the N_{min}-drill within the experimental plots of experiment B. It was not possible to indicate tendencies of pesticide movements into the deeper soil, or the uplifted groundwater, following the analysis of those samples. The vertical sections cut at the N_{min}-drill sampling spots after sampling also indicated, based on Vitasin Blue indications, no or very little water fluxes at the sampling spots.

5.2 COMBINED PESTICIDE-DEUTERIUM TRACING EXPERIMENT

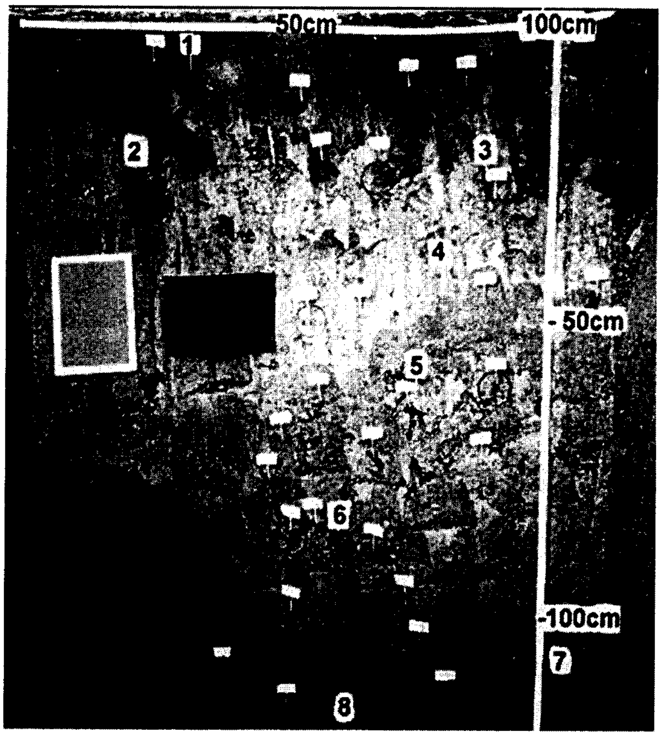
Another experiment C was set up in order to study the water fluxes in the watershed. It was decided to combine the studies in a second phase with flux identifications for pesticides. In this case, a 10 m^2 experimental plot in a *khet* environment was used, which was equipped with suction candles in the depth of 15, 30, 60, 100, 130 and 160 cm below ground. The pesticides were applied with the technique and application amounts identically to experiment type B. Thereafter the plots were flooded with irrigation water enriched with 1 litre of 82% D_2O. The resulting Deuterium breakthrough curves in combination with the pesticide appearances are shown in Figure 8 [6]

Results show a clear indication that pesticides, more especially Dimethoate and Metalaxyl, are transported into the soil through water fluxes, though they do not allow conclusions on the correspondence of pesticide transport on Deuterium breakthrough curves. Fenvalerate again shows interesting transport pattern as it appears in a depth of 160 cm four months after application. Here shall also be noted that samples taken from piezometres within the same test plot have shown considerable amounts of Dimethoate and Metalaxyl one day after application in 45 cm to 145 cm depth (deeper not sampled) [6]. Both results indicate clearly a risk of transportation of the respective pesticides Metalaxyl and Dimethoate, but also Fenvalerate, along preferential flow paths into greater depths reaching down into soil water and to the uplifted groundwater table, thus resulting in a potential pollution risk. Since findings could not be corresponded to the Deuterium breakthrough curves no quantification was possible. Hence, no appropriate risk analysis was carried out.

6. Discussion of results

The environmental risk assessment based on environmental modelling developed by Apel [1] within the research project (see section 4) has served to exclude acute risk of pesticide residue formation in soils or groundwater contamination risk with a probability of >99.5%. It furthermore allowed to name a minimal risk (<1%) of long term residues formation, more especially for synthetic pyrethroids. These findings were backed by soil and groundwater monitoring for pesticides throughout the project duration which did not detect pesticides in any measurable quantities.

The approach followed here, i.e. to formulate a risk assessment for environmental compartments, in this case being soil and water, is certainly not common as risk assessment usually accounts for human beings as the final people at risk. However, being geoscientists we believe that risk to nature also matters and that it may therefore just abstractedly be defined. In a further step though, sensible limits (as indicated before by the quantification limits introduced in section 4) should be defined for pesticide concentrations in the soil. Those sensible limits should not be exceeded in order to minimize residue formation and to ensure that concentrations lie below toxicity for micro-organisms and molluscs.



Sample	Fenvalerate µg/kg	Metalaxyl µg/kg	Dimethoate µg/kg	Vitasin Blue mg/kg	
1	53	62	13	627	
2	14	29	21	135	
3	n.q.	1	n.q.	19	
4	n.d.	2	n.d.	n.d.	
5	n.d.	n.d.	n.d.	<1	
6	n.d.	3	3	24	
7	n.d.	2	7	26	
8	n.d.	n.q.	n.d.	n.d.	
9	n.d.	n.q.	n.d.	n.d.	Sample 9: 1,6 m b.s.
10	n.d.	n.q.	n.d.	n.d.	Sample10: 2,1 m b.s.
11	n.d.	11	11	221	Sample 11: 2,3 m b.s.
12	5	14	9	166	Sample 12: 2,4 m b.s.
a	135	86	>10		
b	n.q.	n.q.	n.d.		
c	n.q.	n.d.	n.d.		
d	n.q.	n.d.	n.d.		

n.d.: not detectable
n.q.: not possible to be quantified

Fig. 7. Soil profile with Vitasin Blue FCF 90 indications [mg/kg soil] and pesticide concentrations [µg/kg soil] after 3 days(Experiment B). Samples 1-12 indicate sampling technique as for experiment type B, while samples a-d (0-5, 5-30, 30-60, 60-90 cm) indicate sampling technique as for experiment type A.

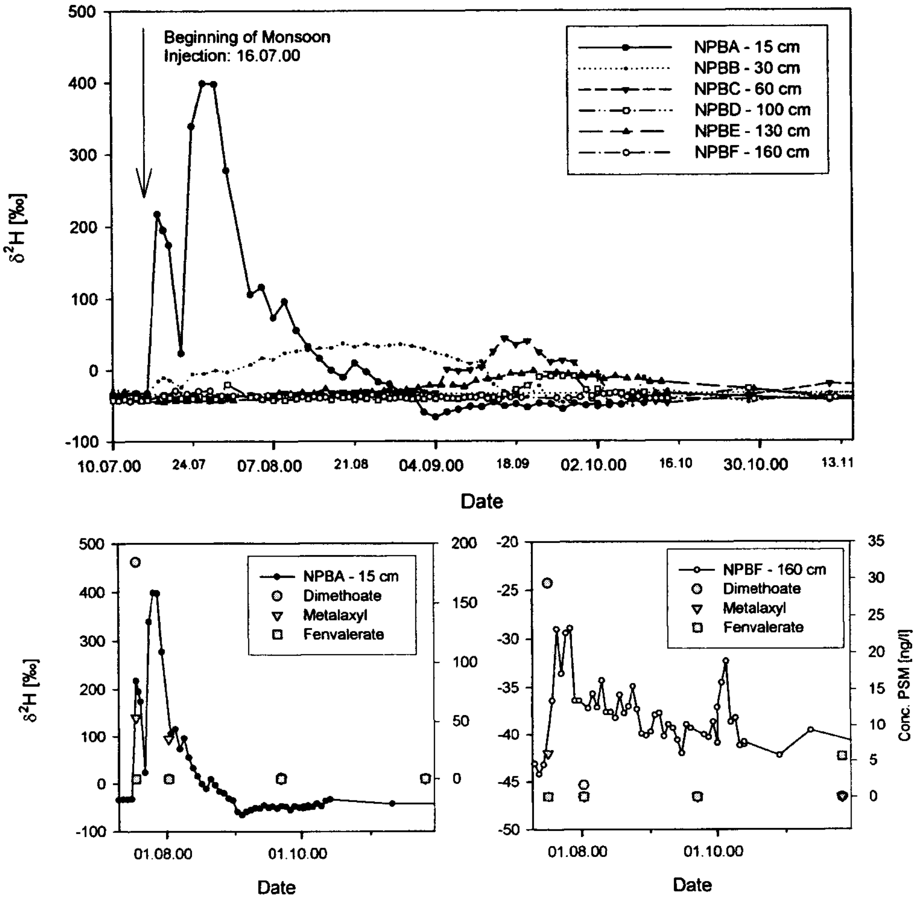


Fig.8. Deuterium (D_2O [‰]) breakthrough curves at plot scale for the experiment started at the beginning of monsoon 2000. Shown are deuterium concentrations in soil water samples extracted through soil suction candles and pesticide appearances. Sampling took place four times. Results for the pesticide concentrations [ng/l] are shown exemplary in the lower plots for soil suction candles installed at 15 cm and 160 cm below surface.(From [6])

Furthermore, the exchange between soil and ground-water could be defined in order to allow a statement on the risk for people through drinking water, by using given limits for pesticide concentrations in drinking water.

The model could not satisfactorily be developed for Fenvalerate, and the assessment of some profiles show phenomena in the data which cannot be explained with the applied model, i.e. higher pesticide concentrations in greater depths than at

medium depth. The detailed transport studies showed that higher pesticide concentrations in greater depth are reality, and that this phenomenon is directly bound to preferential flow paths. The latter allow a fast pesticide transport into greater depths up to the groundwater table, which has been shown not to depend on the pesticides' water solubility. The detailed transport studies have also proven that Fenvalerate, as well as Metalaxyl and Dimethoate, is definitely transported into the ground and that it tends to prolong.

The additional tracing experiments have also shown that it might be questionable whether the sampling technique using an N_{min}-drill (experiment type A) is appropriate as soon as preferential flow paths play an important role for transport mechanism and for the formation of local contaminant pools in the deeper soil. Besides the risk of taking samples always between pathways (this risk might be quantifiable) this sampling technique does also not account for the free soil water. According to the findings of the combined deuterium-pesticide tracing experiment, the free soil water plays a leading transportation role throughout the soil compartment down to the groundwater under Nepalese agricultural conditions. As a next step it would be desirable to link water fluxes and pesticide transport determined through tracing experiments to one-site kinetic sorption models. It can be definitely stated that such detailed transport studies are necessary to reach a sound risk assessment.

Finally needs to be noted, that the investigations concentrated on vertical fluxes of water and pesticides in the soil zone only. Lateral fluxes, which are bound to exist in the studied agro-ecosystem have not been considered here. This is, besides the heterogeneity of soils, and the dependence of pesticide kinetics on temperatures and soil moistures one of the reasons why regionalization of the presented results is not possible.

7. Acknowledgement

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A COMPARATIVE RISK APPROACH TO ASSESSING POINT-OF-USE WATER TREATMENT SYSTEMS IN DEVELOPING COUNTRIES

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Abstract

Unsafe water is a leading cause of death and disease in economically disadvantaged societies. The development of centralized large-scale water treatment and supply systems has proven to be a slow, expensive strategy to provide safe drinking water in many low-income countries. Governments and non-governmental organizations have therefore increasingly been promoting point-of-use water treatment technologies in communities without reliable municipal water supplies. These technologies aim to be low-cost sustainable solutions that rely on filtration, disinfection and safe storage to improve source water quality. This paper uses a comparative risk assessment methodology to quantify the health and water quality impact of a point-of-use water treatment program being implemented in rural Haiti by a non-governmental organization. An observational study was used to measure diarrhea incidence in 120 families in the village of Dumay in Haiti. Univariate and multivariate statistical methods were used to (i) quantify the impact of the water treatment system in reducing the incidence of diarrhea, controlling for socio-economic differences in the population, and (ii) study the interaction of socio-economic factors and source water quality with filter use in diarrhea reduction. As part of the water quality impact assessment study, the microbial content of source water and stored water in intervention and non-intervention households was measured using membrane filtration tests. The comparative risk approach used in this study is designed to provide insights and inputs into environmental decision-making issues relating to resource allocation between competing gastro-intestinal disease reduction initiatives such as point-of-use water treatment systems, high-quality source water development projects, and household safe storage mechanisms.

1. Introduction

Unsafe water is a leading cause of death and disease in economically disadvantaged societies. Over 2 million people die every year of waterborne diseases such as cholera, typhoid fever, amoebic dysentery, and other diarrheal diseases. (Mintz *et al.*, 2001). Despite the horrific human cost of unsafe water, advances in water treatment and supply technologies have been slow to diffuse into low-income countries. Since 1990,

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the number of people without access to clean drinking water has remained nearly constant at approximately 1.1 billion people (WHO/UNICEF/WSSCC, 2000).

Slow economic growth, political instability, social upheavals and poor institutional development in large areas of the developing world suggest that large-scale municipal water treatment and supply systems are unlikely to be established in many urban communities in the near future. In rural areas, large-scale treatment plants may never be economically viable.

Governments and non-governmental organizations have therefore increasingly been promoting point-of-use water treatment technologies in communities without reliable municipal water supplies. These technologies aim to be low-cost sustainable solutions that rely on filtration, disinfection and safe storage to improve source water quality.

Another type of intervention has focused on the development of clean water sources. Well development programs in rural areas, for instance, are aimed at reducing dependence on contaminated surface water sources. Both types of interventions are often used in parallel when even the best available water sources require further treatment.

This paper documents a study that uses a comparative risk assessment approach to evaluate the health and water quality impact of a point-of-use water treatment program being implemented in rural Haiti by a Florida-based non-governmental organization. The study is based on field research conducted in Dumay, Haiti by the author in January 2002 as part of his thesis research at the Massachusetts Institute of Technology towards a Master's degree in Civil and Environmental Engineering.

A number of previous studies have randomly assigned point-of-use water treatment systems amongst treatment and control groups and performed follow-up analyses of the differential health impact in the two groups. Studies have also examined the acceptability of various treatment devices and community compliance rates with treatment methodologies. In these intervention studies, population heterogeneities are dealt with by random selecting the treatment and control groups; treatment groups are provided access to the water treatment system while control groups are expected to continue using their usual water sources. In some studies, extensive education on filter use practices is continually imparted to the treatment group for the duration of the study to ensure compliance with best practices. While such studies provide a good estimate of filter efficacy in an ideal or near-ideal setting, they do not address issues that are of interest in evaluating the field performance of point-of-use water treatment systems in a mature program.

The study described in this paper seeks to provide a snapshot analysis of the efficacy of a representative point-of-use water treatment program that has been operational in a rural setting in a developing country for a number of years. The study seeks to provide answers to the fundamental question of whether filter use is associated with lower incidence of diarrhea disease, all other factors being equal; in addition, the paper seeks to answer vital associated questions such as: How do socio-economic factors like age, education, housing quality and sanitation facilities interact with filter use in reducing diarrhea incidence? In families without filters, how do these socio-economic factors influence diarrhea occurrence? Does the quality of source water

influence gastro-intestinal disease occurrence – how does the impact differ in families with and without filters? Do families using filters have a different profile from families not using filters? How good is compliance with filter use best practices – is this associated with any socio-economic factors? Is post collection contamination a problem in the population? Answers to these questions are likely to afford insight into the most appropriate mechanisms for achieving the goal of gastro-intestinal disease reduction .

In order to investigate these issues, an observational study was conducted on a village in rural Haiti in which the US-based NGO Gift of Water, Inc. has been implementing a point-of-use water treatment program in alliance with local partners for over a decade. Samples were randomly selected from the population using filters and the population not using filters. Since these groups could not be assumed to be homogeneous, information was collected on relevant socio-economic factors for each individual in the two populations. Information was also collected on the incidence of diarrhea in the past four weeks. To study the impact of source water quality on health outcomes and to estimate the severity of post-collection contamination, micro-bacterial tests were conducted on all major water sources in the village, as well as in each household. In filter-using families, in-house tests were conducted to determine if the filter was being used in accordance with best practices. Univariate and multivariate statistical methods were used to (i) quantify the impact of the water treatment system in reducing the incidence of diarrhea, controlling for socio-economic differences in the population, and (ii) to study the interaction of socio-economic factors with filter use in diarrhea reduction. The impact of source water quality on health outcomes was also studied in families with and without filters. The study also examined the role of post-collection practices as a route of water contamination.

2. Background

2.1 HAITI

2.1.1 Geography: The Republic of Haiti is a small, poor, densely-populated country that occupies the rocky western third of the Island of Hispaniola, located between the Caribbean Sea and the North Atlantic Ocean. Its neighbor to the east, the Dominican Republic, occupies the rest of the island.

2.1.2 Human Development Indicators: Haiti ranked a low 134 on the UNDP's human development index, a composite indicator of human progress, in 2001 (UNDP, 2001). Life expectancy at birth in Haiti is 52 years, much lower than the regional average of 70. Adult literacy is only 49%. The infant mortality rate per 1000 live births is 70, more than twice the regional average. The under-5 mortality rate per 1000 live births is 129. Malnutrition affects 28% of children under 5 years. Only 46% of the population has access to an improved water source and only 28% use adequate sanitation facilities. Haiti is one of the most densely populated countries in the region, with a fertility rate of 4.8 compared to the regional average of 2.8. Haiti's population was estimated at 8 million in 2000.

2.1.3 Economy: Haiti is the one of the poorest countries in the Western Hemisphere, with a gross national income per capita of US $ 480 (Atlas method) in 2000 (World Bank, 2001). This compares unfavorably with even sub-Saharan countries and is much lower than Haiti's neighbors in Latin America and the Caribbean. The total gross domestic product was only $4.3 billion in 1999. The economy steadily stagnated in the past decade with a per capita GDP growth of -3.4% in 1990-99. The agricultural sector accounted for a little under 30% of gross national product, with industry and services accounting for 21% and 49% respectively in 2000.

2.1.4 Climate and Environment: Haiti experiences tropical climatic conditions except in the semi-arid east, where mountains cut off the trade winds. The terrain is mostly rough and mountainous. The island lies in the middle of the hurricane belt and is vulnerable to severe storms from June to October. The country has suffered extensive deforestation and soil erosion, particularly after US sanctions in the 1990s. Much of the remaining forest is being cleared for agriculture and fuel.

2.1.5 Intestinal Infectious Diseases: Diarrheal disease was the leading cause of illness and death in children under 5 years of age in the 1990s (PAHO, 1998). The incidence of diarrhea in the general population was as high as 47.7%, according to health surveys conducted between 1987 and 1994. Typhoid is endemic in Haiti. It ranked as the fifth leading cause of hospitalization during some periods of the 1990s.

2.2 GIFT OF WATER, INC.

Gift of Water, Inc. (GWI) is a public charity based in Brevard County, Florida. Its mission is "to provide clean drinking water and community development to the impoverished people of developing countries through the use of home-based, appropriate technology water purifiers." (Www.giftofwater.org)

GWI initiated its activities in Haiti in 1995 and currently operates water treatment programs in seven different communities across Haiti. The charity works with various church-based organizations and aims to "meet not only physical but also spiritual needs of the disadvantaged" although it "strives to be undiscriminating in the communities it helps." GWI has distributed approximately 3,000 filtration systems amongst these seven communities.

2.3 DUMAY

The site of GWI's first water treatment intervention in Haiti, Dumay is a cluster of villages located approximately 15 kilometers south of Port-au-Prince. The study documented in this thesis was conducted mainly in Dumay. About 5% of the data was gathered from two small villages called Bonnette and Beauge, which are located approximately five kilometers outside of Dumay.

Dumay is a flat plain ringed by hills that are largely bare of vegetation. Deforestation and erosion have exposed stony white swathes in some elevated areas. The plain below is still green and mostly cultivated. Several springs and a small river flow through the land. Some of these surface water sources are channeled for irrigation.

Principal crops grown in the area include sugarcane, sweet potatoes, corn, beans, tomatoes and vegetables. The villages are spread over an area of approximately 15 square kilometers, mostly comprising cultivated fields. The roads connecting the village clusters are unpaved. Private operators who drive jeeps called tap-taps provide intermittent public transport.

The villages are named Haut Campeche, Bas Campeche, Celicourt, Temoulin, Tijardin, Barriere Rouge, Lorial, Barriere L'Hopital, Denis, Maroseau, Jean Mary, La Hatte, Liziere, Delmas, Turbe, Jonc, Drouillard, Barron, Gamant, Boiscabrit, Pierroux Douceur, Pont Dumay, Galette Dumay, Pernier, Carrefour Pernier, Timoulin, Bambour, Galette Drouillard, Terresalee, Digneron, Rocheblanche, Coupont, Guedon, Duval Amboise, Laferme, Michaud, Haut Cottard, Laferronnee, Trois Rigeoles and Noailles. These names correspond to different areas of what may be considered one large village.

There is no public hospital in Dumay, but some churches hold occasional medical clinics. There are numerous churches in the village. Four major schools affiliated to local parishes serve the area. Much economic activity in Dumay is agriculture-related. Nearly all the households surveyed worked the land, either in a share-cropping arrangement or as agricultural labourers. A few people owned land and hired labor to cultivate it. Goods are traded at a weekly market. A number of women worked part-time as vendors. A variety of farm animals are raised including cows, goats, sheep, pigs, chickens, ducks, turkeys, donkeys and even horses.

Although agriculture accounted for most employment in Dumay, there was considerable divergence in household wealth and quality of housing. This may be attributed to skewed land distribution and expatriate incomes from relatives working in the United States in some families. The community did not show large divergence in educational attainment. The average family education deficit, defined as the difference between ideal and actual educational attainment for age, was 6.5 years with a standard deviation of 2.8 years. The study found only 4 university graduates amongst 841 individuals. The survey indicated that the majority of the population in Dumay was Protestant. About 60% of the survey population was Protestant and a little over 30% was Catholic.

Most families used one of two main water sources. (i) Piped spring water capped at the source of springs in the surrounding hills and available at common village taps. (ii) Hand-pumped tube wells constructed by non-governmental organizations.

2.4 THE GWI FILTER

GWI's current chlorine-based purifier design comprises two detachable 19-litre plastic buckets connected by a check-valve. Users fill the top bucket with water, add a 5 ml dose of 5.25% sodium hypochlorite solution, and allow the water to stand for 30 minutes. This contact time with chlorine is expected to kill bacteria and viruses in the water. At the end of 30 minutes, the top bucket is lifted onto the check valve fitted to the bottom bucket, which starts flow into the lower bucket. Water flows through a polypropylene sediment filter in the top bucket and into the bottom bucket through a granular activated carbon (GAC) filter. The GAC removes the chlorine and many other chemicals that might be present in the water. A spigot on the bottom allows users to

draw clean water directly from the purifier. Five drops of residual chlorine added to the bottom bucket prevent pathogen regrowth during storage.

The filters are currently manufactured in the United States and assembled in Haiti. The filters cost US $15 and cost recovery is currently US $2.

3. Survey Design and Sampling Methodology

A total of 120 families were surveyed in Dumay, 62 of whom owned a GWI filter. The program did not cover the remaining 58 families; these families did not regularly use any water treatment system. Filter-owning households were randomly selected from GWI's program records using a random number generator. The GWI program distributes its filters by a means of a promotion drive in each of several circuits in the village at the end of which interested families are required to visit the GWI program office and apply for a filter. Each circuit, corresponding to administrative sub-divisions within the village, was sampled in proportion to its representation in GWI's program.

In the absence of land ownership or census records, non-filter owning households were selected using the following ad-hoc randomization algorithm. From each sampled filter-owning household, a random number generator was used to pick a number between 1 and 15, which represented the number of houses to skip before sampling a fresh house, and a number between 1 and 4, which represented the direction in which to proceed. If the process led to a household that either used a filter or that had previously been sampled, the algorithm would be repeated from that point. This is clearly a non-ideal randomization process that would be inadequate if there were pockets of non-filter owning households far removed from filter-owning households. However, interviews with the staff of GWI and personal observations indicated that the two populations were well mixed and did not show any systematic geographic separation.

4. Survey Data Collection

One respondent in each family was administered a survey that solicited information on the family's socio-economic status. The respondent was in most cases the mother or grandmother of the family. If other members of the family were present at the time of the survey, which was frequently the case, each individual member was asked about health status in past month, specifically incidence of diarrhea and bouts of fever. Diarrhea was defined as three or more incidences of watery stools in a day. The respondent was also asked to volunteer information about the health status of all family members that she had accurate knowledge about. In the event of contradiction, the respondent's version was accepted over that of a child and very old persons. If any family member was absent, the respondent was asked for information regarding his or her health status in the period of interest; if the respondent was unaware of the health status of the absent individual in the period of interest, that individual's status was marked as "not available."

An attempt was made to emphasize the period of interest (one month) by referring to a calendar, if there was one present in the home, or asking the respondent/s to relate the period of one month to the past four church services in the village, or to the period elapsed since Christmas. This was facilitated by the fact that Christmas, which is celebrated by nearly all people in the village, was between 3 weeks to one month prior to the time of administration of the survey. The survey was conducted in Creole, the language most widely spoken in rural Haiti, by a senior GWI technician, in the presence of the author. Survey responses were recorded in English after simultaneous translation.

Each family was requested to be as accurate in their responses as possible. However, the accuracy of some responses could possibly have been compromised by the reluctance of respondents to speak openly of their illnesses in front of the GWI technician, a well-regarded community member. The incidence of HIV seropositivity is around 5% in rural Haiti and families may have feared that admitting to illness might be construed as evidence of HIV. However, there are indications that there may have been no systematic under-reporting of diarrhea, since many more people reported incidence of fever than diarrhea. Given that fever is well known to be a symptom of HIV/AIDS in the community (according to a local health worker), it seems unlikely that a family would admit to fever and not to diarrhea to prevent an adverse assumption of HIV infection.

Some families may also have construed the survey as a test to determine whether they ought to receive a filter. As a consequence, respondents may have tried to provide "ideal" responses to questions on behavioral issues and possibly even on health status. Another data quality problem relates to the accuracy of health data gathered from one respondent about other family members, particularly for a duration of one month. It is quite possible respondents did not have perfect knowledge of diarrhea episodes experienced by all members of the family, or that they did not precisely remember whether the episode had occurred in the past month. Given the proximate living conditions in Dumay, where families of 8 or more persons often shared just one or two rooms, it is likely that information gathered from mothers and grandmothers about the health status of their children would be accurate. Cross-checking with the individual, most of who were present during the survey would also have enhanced accuracy. The recall period conveniently occurred approximately between Christmas day, a well-remembered day in the local population, and the administration of the survey, which took place between Jan 17 and Jan 25.

Respondents in filter-owning households were also surveyed on how frequently they drank water from other non-filtered sources. This question appeared to receive the most "model" answers and was disregarded in the data analysis.

In summary, the analysis compares reported health outcomes in individuals from filter-owning households and non-filter owning households, without any adjustments for the possibility that (i) the reported outcomes may be inaccurate or (ii) individuals from filter-owning households may occasionally be drinking non-filtered water.

5. Survey Variables

Apart from health outcomes, information was gathered on all measurable variables that were likely correlated with health outcomes. In particular, those variables were included that are likely correlated with both health outcomes and inclusion in the GWI program. For instances, income and education are variables likely to be correlated with inclusion in the program (wealthier and educated people are more likely to have filters) as well as health outcomes (wealthier and educated people are likely to be healthier). Excluding control for income and education would thus exaggerate the impact of having the filter on health outcomes.

The survey gathered information on the following variables and groups of variables:

1. Health Outcomes: (i) A binary variable for whether or not each member of the family had experienced diarrhea (defined as three or more loose stools in a 24-hour period) in the past month. (ii) A binary variable for whether or not each member of the family had experienced fever (as defined by the family) in the past month.
2. Filter: a binary variable for whether or not the family used a filter.
3. Geographic Location: A category variable for the circuit number assigned by GWI to each area, which roughly corresponds to a village unit.
4. Household Size: A continuous variable for the number of persons living in the surveyed house.
5. Source of Water: A category variable for source of water
6. Sanitation Facilities: A category variable that measured whether a family used (i) a private bathroom (ii) a common bathroom or (iii) no sanitation facilities.
7. Quality of Housing: A category variable that classified houses as: (i) Earthen walls and floor, corrugated iron roof (ii) Earthen walls, cement floor, corrugated iron roof (iii) Cement walls, floor and roof, unpainted, unfinished fittings (iv) Cement walls, floor and roof; partially finished fittings (v) Completed concrete structure with modern fittings.
8. Rooms: A continuous variable for the number of rooms in the house
9. Electricity: A category variable for (i) No electricity (ii) Illegal Connection (iii) Legal Connection (iv) Generator
10. Age: A continuous variable for the age of each family member in years.
11. Education: A continuous variable for the educational attainment of each family member
12. Occupation: A category variable for the occupation of each family member, recorded as (i) Share-cropping (ii) Cultivation of own land with hired labor (iii) Agricultural labor (iv) combination of share-cropping and agricultural labor (v) Services (Mason, Driver, Mechanic, Pastor, Bicycle Mechanic, Cook, Teacher) (vi) Factory Worker (vii) Vendor (viii) Commercial Enterprise (ix) Transfer from Family Member and (x) Professional Services (Lawyer, Nurse)
13. Religion: A category variable recorded as (i) Catholic (ii) Protestant (iii) Voodoo (iv) no reported religion.

14. Family Assets: Continuous variables for the number of assets such as cows, goats, chickens, pigs, donkeys, sheep, horses, ducks, cars, TVs, radios and luxury appliances.
15. Behavioral Characteristics: Category variables for information on use of soap, diapers and hand-washing habits.
16. Use of other water treatment systems: a category variable recorded as (i) Add chlorine sometimes (ii) Borrow filtered water from friends when sick (iii) Boil water when sick (v) Boil water always.
17. Filter use characteristics: Continuous variables for the date of last cleaning, year installed and rate of use.
18. Reason no filter: A category variable for the reason families did not own filters.
19. Household hygiene: An observational measure of hygiene on a scale from 1 to 5.

Inclusion of the entire vector of survey variables in a statistical analysis procedure such as a regression would create problems with multicollinearity, owing to the inter-related nature of the variables, and result in statistically insignificant results given the parsimonious sample size. Nor would inclusion of individual variables such as the number of donkeys owned by a family be particularly meaningful or helpful in explaining differential health impacts. It was therefore necessary to collapse the extensive vector of survey variables into composite variables that compressed relevant information into a more useful form.

1. Composite Wealth Variable: Family assets were multiplied with average asset values to generate a cumulative asset value of observable family assets for each family. However, this cumulative variable is incomplete as a measure of family wealth as it omits assets such as bank accounts and expatriate incomes from family members in the United States. This variable was therefore combined with the quality of housing variable to produce a categorical measure of relative wealth ranging from 1 to 5, weighting quality of housing by 75% and the cumulative asset value variable by 25%. A higher weight was accorded to the quality of housing variable, as health outcomes are more likely to be influenced by investments in housing than in the other non-visible assets that might have been missed in the cumulative asset value variable.
2. Composite Hygiene Variable: The observational hygiene variable was combined with the see-soap variable, which checked whether the household had soap. A higher weight was placed on the observational hygiene variable since the absence of soap might have indicated a temporary unavailability.
3. Family Educational Deficit: In considering the impact of educational attainment on health outcomes, the average family education deficit was considered as a relevant explanatory variable rather than an individual's education level. This captures the impact of parental educational levels on family health outcomes. Each individual's deficit was worked out as the difference between the educational level the individual should have attained for her age with no breaks in the education process and the individual's

attained level. The optimal level for an adult was defined as a university degree.

4. Adult Education Deficit: Since children contribute little to the educational deficit, families with high numbers of children register a low average family educational deficit. The adult education deficit improves on the average family measure by considering only the average of the education deficit of persons above 15 years of age. This would include the members of the family mainly involved in household tasks such as cooking, cleaning and looking after children and other tasks in which educational attainment can be expected to improve hygiene practices relevant to health outcomes.

6. Statistical Methods

Exploratory univariate analyses were performed to evaluate trends between the incidence of diarrhea and various socio-economic factors, and between filter ownership and the same socio-economic factors. The factors that were found to be jointly significant in influencing the incidence of diarrhea and filter-ownership were selected for analysis in a multivariate framework.

6.1 UNIVARIATE ANALYSIS

Pearson chi-square tests, Fisher tests, analysis of variance, and t-tests were used as appropriate to determine associations between the incidence of diarrhea and the entire range of socio-economic and water quality indicators on which data was gathered. The same tests were used to assess associations between filter ownership and the socio-economic and water quality variables.

6.2 MULTIVARIATE ANALYSIS:

Ordinary least squares regression, probit regression and logistic regression were used to analyze the impact of filter-ownership on the incidence of diarrhea, after controlling for all socio-economic and water quality variables that were found to jointly associated with diarrhea incidence in the univariate analysis. Interaction variables between the socio-economic variables and filter ownership were also included. Quadratic terms were assessed for statistical significance and were found to be significant only for the age variable.

7. Summary Of Key Results

7.1 HEALTH IMPACT

- Of the total sample of 841 individuals, 86 (10.23%) were reported to have experienced at least one episode of diarrhea in the month preceding the survey.

- Of the 380 individuals with no access to a filter at home, 56 (14.74%) were reported to have experienced diarrhea in the month preceding the survey.
- Of the 461 individuals with access to a filter at home, only 30 (6.51%) were reported to have experienced diarrhea in the month preceding the survey.
- The population with access to a filter therefore experienced an 8.23 percentage point, or 56%, lower incidence of diarrhea than the population with filters. The result is statistically significant at $p = 0.0001$.
- Intervention and non-intervention populations could not be considered identical.
- Intervention households, who owned a filter, had a higher average quality of housing, better sanitation facilities, larger household size and lower education deficit than the non-intervention households.
- After controlling for divergences in socio-economic factors and observed hygiene, the filter was associated with a 5.2 percentage point lower probability of diarrhea. The result is statistically significant at 95% confidence.
- Improved sanitation facilities were independently associated with lower diarrhea incidence at 95% confidence.
- Improved quality of housing was independently associated with lower diarrhea incidence, but the result was not statistically significant at 95% confidence.
- Better hygiene was associated with lower diarrhea incidence, but the result was not statistically significant at 95% confidence.
- Age was closely correlated with diarrhea incidence, with younger and older persons more at risk. The result was statistically significant at 95% confidence.
- For children of age 5 and under, the filter was associated with a lower diarrhea probability of 16 percentage points, controlling for all other factors. The average incidence of diarrhea in the group was 31.16%.
- For older children in the 6-16 age group, the filter was associated with a lower diarrhea probability of 4 percentage points, controlling for all other factors. The result was not significant at 95% confidence.
- For persons of age 16 and older, the filter was associated with a 4-percentage point lower probability of diarrhea, controlling for all other factors. The result was not significant at 95% confidence.
- Of the 369 individuals who used piped spring water, 11.38% experienced a diarrhea episode.
- Of the 332 individuals who used well water, only 7.23% experienced a diarrhea episode.
- Amongst the population that used piped spring water, the filter reduced the incidence of diarrhea by 7.7 percentage points.
- Amongst the population who used well water, the impact of the filter was not significant in explaining diarrhea incidence.

7.2 WATER QUALITY IMPACT

7.2.1 Source Water Quality

- The average total coliform count in duplicate samples from 23 wells was 2.65 cfu/100 ml.
- The average *E. coli* count in duplicate samples from 23 wells was 0.28 cfu/100 ml.
- The average total coliform count in duplicate samples from 3 community taps that were supplied with piped spring water was 214 cfu/100 ml.
- The average *E. coli* count in duplicate samples from 3 community taps that were supplied with piped spring water was 3.5 cfu/100 ml.

7.2.2 Stored Water Quality in Intervention Households

- In 11 filter-owning households with adequate residual levels of chlorine in both buckets, there was little evidence of microbial contamination, with average TC= 0.68 cfu/100ml and average EC=0.05 cfu/100ml.
- In 10 filter-owning households with adequate residual levels of chlorine in the top bucket but inadequate chlorine in the bottom bucket, there was little evidence of microbial contamination, with average TC= 2.25 cfu/100ml and average EC=0.1 cfu/100ml.
- In 1 filter-owning households with inadequate residual levels of chlorine in the top bucket but adequate chlorine in the bottom bucket, there was little evidence of microbial contamination, with average TC= 10.5 cfu/100ml and average EC=0 cfu/100ml.
- In 10 filter-owning households with inadequate residual levels of chlorine in both buckets, there was substantial evidence of microbial contamination, with average TC=913 cfu/100ml and average EC=22.3 cfu/100ml.

7.2.3 Stored Water Quality in Non-Intervention Households

- In 42 non filter-owning families, there was evidence of considerable microbial contamination in stored water, with average TC = 2,484 cfu/100ml. In 45 non-filter owning families, the average EC was found to be 162 cfu/100ml.
- Contamination levels were negatively associated with improved sanitation at statistically significant levels for both indicators.

8. Discussion

The case study documented in this paper uses a comparative risk approach to investigate the impact of a chemical disinfectant-based point-of-use water treatment system in a rural area of Haiti. The study found that the intervention population experienced a lower incidence of diarrhea than the non-intervention population. Although part of this effect is attributable to better housing and sanitation facilities, the filter itself was independently associated with a 50% lower incidence of diarrhea. The average incidence rate in the combined sample was approximately 10.5%. The greatest

impact of the filter was felt in two populations: the under-5 age group, in which the filter was associated with a 16 percentage lower probability of diarrhea, and the population that used piped spring water. The impact of the filter on older populations was of a lower magnitude and was not statistically significant. The filter was not significantly associated with lower diarrhea incidence in the population that drew water from wells. Diarrhea incidence was not significantly correlated with an individual's sex, household size or adult educational attainment.

Source water quality analysis corroborated the findings of the health impact study. Water from wells was found to be largely free of faecal contamination. Piped spring water was slightly more contaminated in comparison. Importantly, stored water in non-intervention houses was considerably more contaminated than both types of source water, suggesting post-collection contamination is a major concern. Stored water in compliant intervention households was very pure.

The results of the comparative risk assessment undertaken in this study may provide insight into environmental decision-making questions relating to resource allocation between competing gastro-intestinal disease reduction initiatives such as point-of-use water treatment systems, provision of high quality source water, and stored water contamination reduction. Further investigation of water quality and diarrhea disease incidence trends at different times of the year may provide the basis for effective program restructuring. If well water is found to be of acceptable quality throughout the year (in terms of microbial contamination as well as potability indicators such as salinity and turbidity), GWI may consider encouraging the use of wells over piped spring water in Dumay. This may require a preliminary investigation of groundwater reserves followed by the development of new wells in areas currently served only by spring water. Access to pure water sources would need to be supplemented by interventions that promote the use of safe storage containers and behavioral change on the lines of the CDC's safe water system. In other words, it may no longer be necessary to promote GWI's two-bucket cotton-and-carbon filter system if access to clean source water can be extended to the whole community. Instead, the community could be provided with, or encouraged to buy, two special plastic buckets with a tap and a recessed opening that prevents the entry of hands and other large objects. The stored water would need to be dosed with a limited amount of disinfectant to prevent bacterial regrowth. This approach has the advantage of being cheaper and therefore more easily extended to the wider community. It would also reduce the community's exposure to potentially dangerous tri-halo methanes and other disinfection by-products, currently a concern with the filter.

This study examined only one village. It may not be possible to develop access to clean water sources in all the areas that GWI operates in, owing to financial constraints and/or unavailability of clean sources. GWI's current point-of-use filtration system is an appropriate intervention where clean water sources are unavailable. However, even in such communities, there is a strong case for extending safe storage interventions to the households that are yet to be provided with a filter.

The study also revealed that most diarrhea incidence occurred in the under-6 age group. This suggests that appropriate interventions aimed at families with children in this age group may be an effective means of reducing the overall incidence of diarrheal diseases. Clearly, such program modifications would require considerable

research into economic feasibility and social acceptability, as well as organizational and sustainability challenges. This study does not examine these issues; it merely draws attention to evidence that a point-of-use water treatment intervention based on safe storage, minimal disinfection and behavioral change may be a cost-effective means of increasing program impact in areas with access to relatively safe source water.

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RISK-BASED EVALUATION OF THE SURFACE COVER TECHNOLOGY OF A RED SLUDGE WASTE DISPOSAL SITE IN HUNGARY

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Abstract

During past decades, several million tons of aluminum industry wastes have been dumped, without a bottom lining system, right on the shore of the Danube River in northwest Hungary. The resulting red sludge ponds pose two major threats to the environment: drinking water resources could be polluted by infiltration, and wind erosion contaminated surrounding residential areas. Due to limited resources, the latter issue was not addressed until the present owner and operator of the industrial site took charge. . To solve the problem of cost, the operator applied treated hazardous waste to form a surface cover for the already polluted site. His proposal to he use this technology at the site obtained all the necessary permissions for implementation. Due to a new piece of national regulation, risk-based evaluation of the remediation technology became required. The task of the risk assessment was to compare the original (uncovered) state of the site and its human health risk to the remediated (covered) situation, where the covering layer consisted of manipulated hazardous waste material. This interesting case study of risk analysis, beside its challenge to the assessor's expertise, raised several questions regarding environmental ethics, environmental stewardship, and responsibilities.

1. Introduction

Almásfüzitő is located on the shore of the Danube River, in northwest Hungary near where the river runs on the borderline of Hungary and Slovakia. The area, having been affected by Hungary's aluminum industry for several decades, is heavily industrialized. In addition to the air contamination due to the aluminum preprocessing and smelters, the typical waste of the aluminum industry—the so-called "red mud"—was disposed in large quantities near the populated area (see Figure 1).

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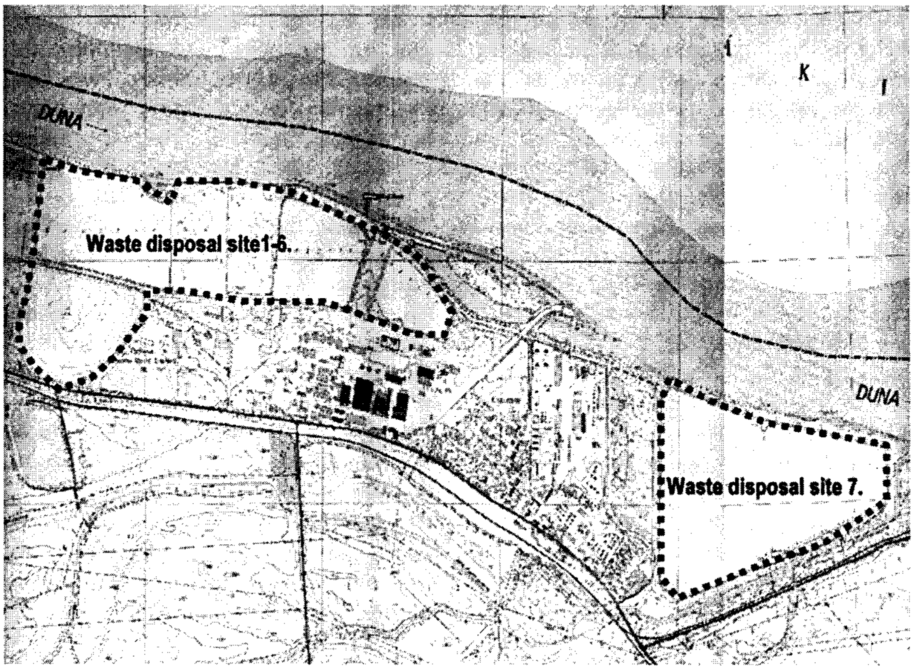

Fig 1. Location of the Almásfüzitő waste disposal site

The waste material is stored in seven waste ponds of varying size and depth. Six disposal sites lie close together, while the last (No. 7), having the largest extent, is located apart from the others. The total amount of waste dumped around the processing plant is approximately 8 million m^3, located on an area of 160 acres. All disposal ponds are located within a couple of meters of the Danube River.

The waste material was transported from the plant to the disposal place in suspended form, through pipelines. The waste was then deposited in the pond, and the excess water was recycled to the processing plant.

The geological setting of the area is a typical alluvial series. The top of the gravel terrace of the Danube is 5-15 meters below the ground's surface. The covering silt, sandy and clayey sediment series have varying impermeability, and cannot be considered by any means as reliable natural liners for such disposal ponds. One typical cross-section, perpendicular to the Danube River, is shown on Figure 2. The measured hydraulic conductivity of the sediments is as follows: sandy clay, $k=5,6.10^{-7}$ m/s; sand, $k=1,4.10^{-4}$ m/s; sandy gravel, $k=5,9.10^{-4}$ - $5,4.10^{-3}$ m/s; gravel, $k=6,7.10^{-3}$ m/s.

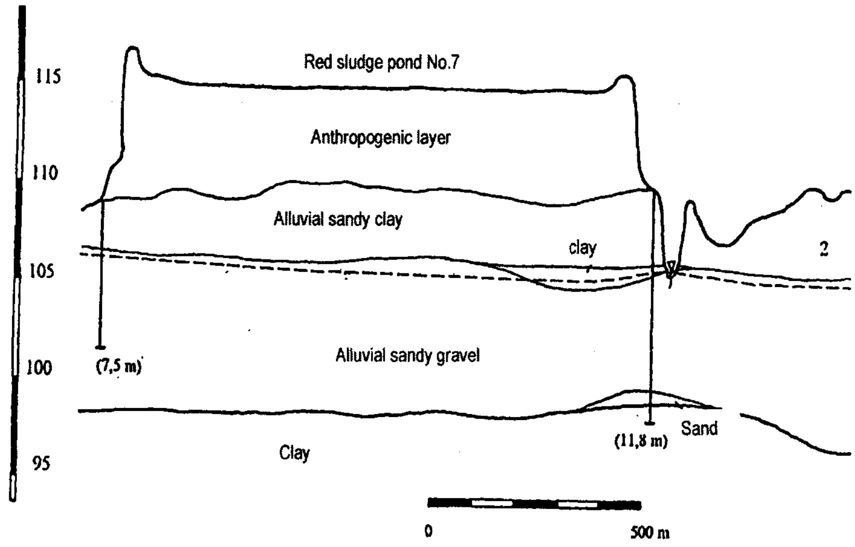

Fig 2. Cross-section of disposal site No7.

The cross-section from disposal site No. 7 shows that the site is sided by the embankment of the Danube on the North and a small canal on the South. The groundwater level of the area is in close correspondence with the Danube's level. Thus, during times when the Danube's waters are high, its piezometric head cuts into the bottom of the waste body. The Szőny-Füzitői canal, which travels parallel with the Danube for several kilometers and passes by the settlements of Szőny and Almásfüzitő, runs into the Danube River at a location bordering the No7. waste disposal pond. During high waters, the Danube's waters raise the water levels in the canal, and its embankments form a pond around an artificial island made of the red sludge in the area. There is no record of installation of bottom-lining systems on the site, which most likely reflects the true situation (no linings existing). HDPE lining was installed only on the embankment sides to strengthen the embankments during site construction.

2. The Red Mud Waste Material

The "red sludge" is a material rich in a great variety of metals, among them heavy metals and other toxic chemicals. Its chemical composition varies from place to place, but several contaminants (As, Ba, Pb, Se, Ni, Cd, Cr) were detected over regulatory limits both on site and in soil samples from adjacent locations.

The polluting effect of the disposal site on the environment is indicated by the soil pH anomalies that originate from the waste material. The pH of the red mud ranges

from 8-11. It is clear that during the last decades an extensive heavy metal pollution of groundwater and soil contamination occurred near the site and it is reasonable to believe that the Danube and the Szőny-Füzitői canal together play a significant role in diluting, draining, and dispersing the contaminated groundwater. Samples of 5 surrounding monitoring wells showed pH levels exceeded 10 occasionally showing a clear effect of the disposal site on the local groundwater. In some places in the vicinity of the disposal sites, the presence of vegetation adapted to highly saline environments also gives additional evidence of the soil contamination and its effects.

The waste material is a thyxotrope sludge having a water content ranging between 76-79%. It is high in clay mineral content; thus, its hydraulic conductivity is 10^{-8}-10^{-9}m/s. After disposal, a 10-30-cm-deep solid crust forms on its surface, and this crust became the source of another environmental problem at the Danube site. The huge, elevated surface areas were swept by winds, which carried away small dried particles containing toxic materials. This toxic dust caused serious air quality problems that threatened the surrounding communities and made vineyards and agricultural lands unusable. As a result, the area was categorized as heavily polluted by the environmental authorities. The measured dust concentrations in the area varied between 50 and 4000$\mu g/m^3$, almost always 10-60% above limits. The magnitude of this latter problem demanded urgent intervention, even though the costs of groundwater remediation exceeded the magnitude of any domestic funds.

3. Surface Covering of the Site

By the early 1990s, the collapse of the industry in Eastern Europe left environmental problems derelict on an enormous scale, and the Almásfüzitő site was one of them. The site, including its environmental legacies, was privatized by a domestic company that undertook the obligation to cover the surfaces of the waste ponds. By this time the company had access to a patented, economically viable surface cover technology that would eliminate wind erosion.

The surface cover technology was a controlled mixture of coal power plant bottom ash and various other hazardous wastes. The mixture was laid on the surface of the red sludge at a thickness of 1 meter and vegetation was planted on the surface. The applicable hazardous wastes were restricted to those where the maximum 10:1 (ash:waste) mixture concentrations (for any contaminant) did not exceed that of the domestic regulatory limit of sludge disposal on agricultural land. The majority of the waste material was hydrocarbon-contaminated sludge, including wastes from the machinery industry, oil-contaminated soil, sludge from the paint and coating industries, etc.

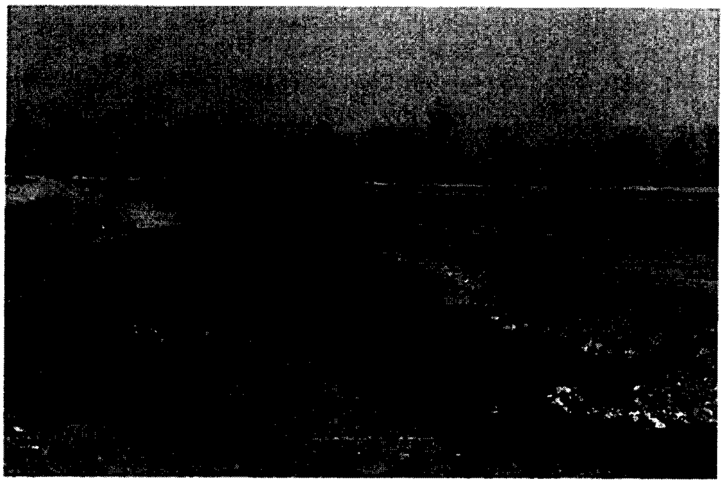

Fig 3. Surface covering in progress at the disposal site

Due to the lack of government financing, the company financed the surface cover implementation using the disposal fee received from the prior owners of the hazardous waste material. The activity was monitored and licensed by the Hungarian Environmental Inspectorate. By 2001, most of the pond surface was covered, eliminating the pollution derived from wind erosion. At that time, new Hungarian regulations came into effect that obliged the site owner to perform a risk-based reevaluation of the technology. The risk assessment of the site was part of a larger site assessment and environmental audit conducted in collaboration with Naturaqua Ltd. (Budapest, Hungary) and Geonardo Ltd (Budapest, Hungary). The risk assessment project was supported by on-site testing and hydrodynamic and transport modeling. The goal of our assessment was not the overall risk estimation of the complex environmental threats posed by the site as a whole, but rather an acceptability assessment of the surface cover technology.

4. Risk Assessment Concept

After coming to an understanding the complex problem and investigating the involved scenarios, the risk assessment concept was summarized in two major questions:

1. Is the new risk component (disposal of new hazardous material onsite) acceptable?
2. Is the resultant change of the overall risk level (before and after surface cover) positive or negative? ($\Sigma R_{before} <> \Sigma R_{after}$)

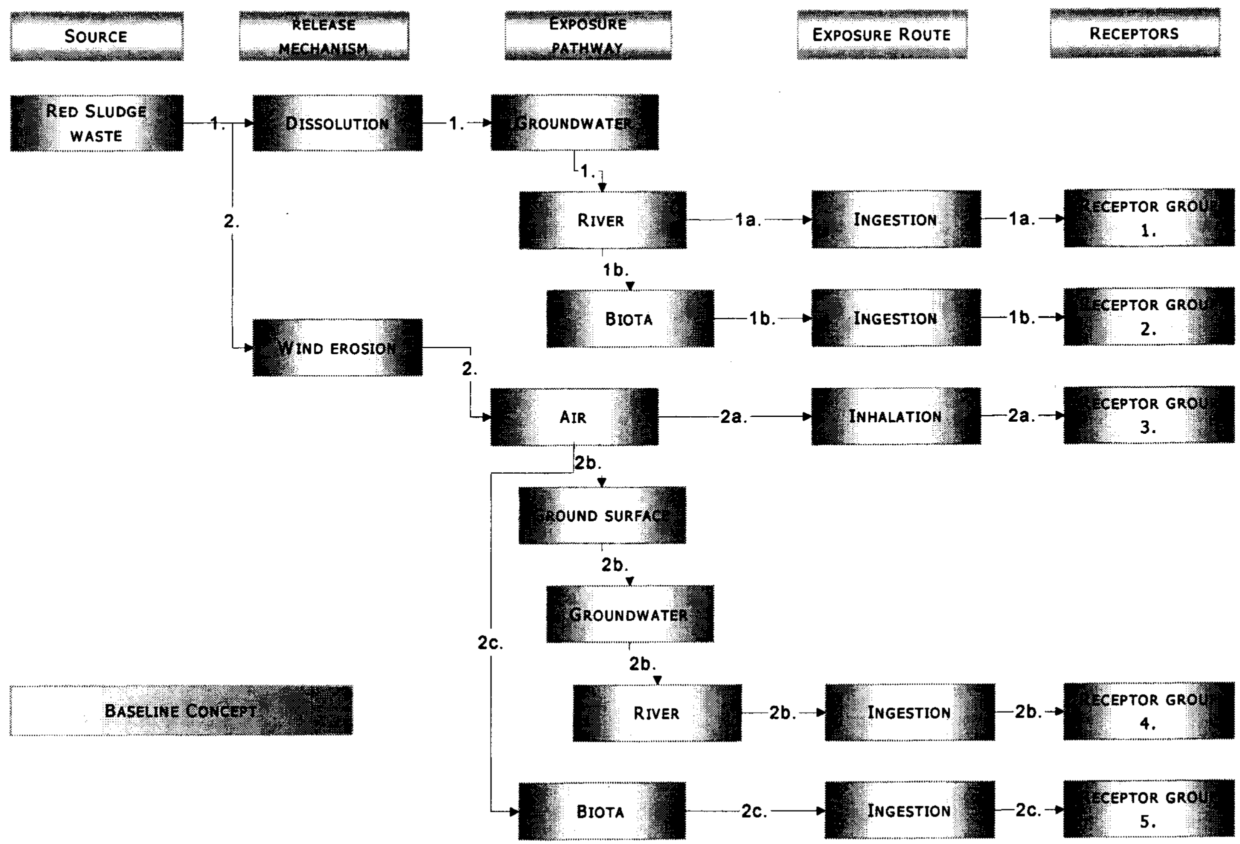

Fig 4. Conceptual model of the red sludge waste disposal site risk assessment

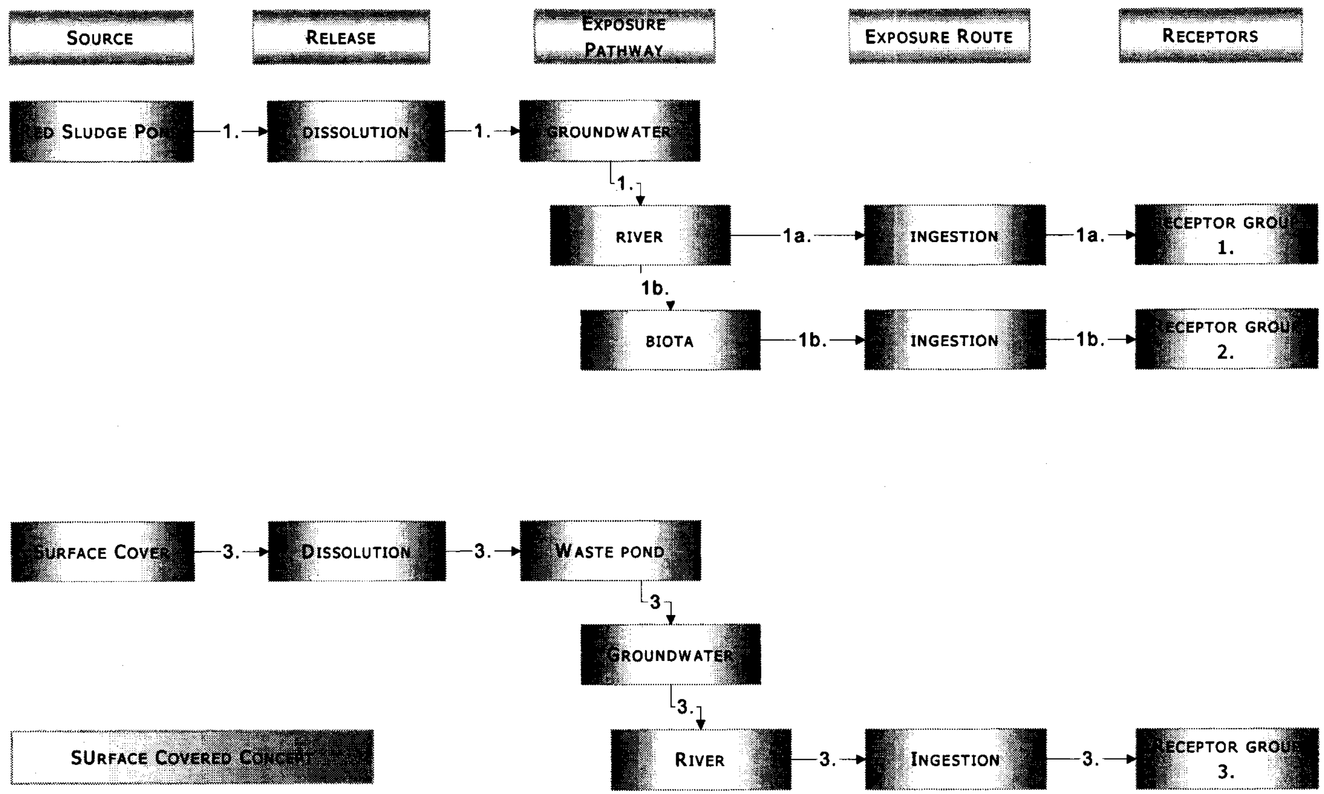

Fig 5. Conceptual model for the risk assessment of the covered waste disposal site

In our assessment, two cases were evaluated and compared. Figure 4 shows the conceptual model of the site before the surface cover was implemented. In this case, two major environmental scenarios needed special attention: groundwater contamination and wind erosion.

$$\Sigma R_{before} = R_{gw} + R_{air}$$

In both scenarios, the toxic components of the red sludge are the sources of contaminants.

In the second case, the surface covering and planted vegetation eliminates the risks from wind erosion, while the groundwater scenario was assumed to remain untouched. In addition, new toxic chemicals were placed on the top of the existing waste sites, from which contaminants may have been transported down to the ground, forming a new pathway of potential threats (see Figure 5). The total risk for this second case is the sum of these two scenarios.

$$\Sigma R_{after} = R_{gw} + R_{surfcov}$$

Assuming that the risk of the red sludge posed by the groundwater is constant for both cases, the overall risk balance can be investigated by comparing the risks posed by the dust erosion of the red sludge to the risk of the new hazardous waste material disposed on the site:

$$R_{air} <> R_{surfcov}$$

Note that in our risk assessment, we have not taken into account the risks of the waste being treated in other place and technology.

According to these observations, our risk assessment concept had the following premises:

The risks posed by the red sludge through the groundwater-surface water are not the subject of the risk assessment; we assume that this risk component does not change due to the surface covering. We also stated our opinion that the risks posed by the red sludge ponds existing, covered or not, on the banks of the Danube poses a much greater risk to a much wider range of receptors than the wind erosion. However, the scale of the remediation and removal for this larger problem is so huge that such an action is not a realistic likelihood, due to the lack of available financing.

The risks caused by particles eroded by the wind must be assessed to evaluate the change in the resultant risk. We also assumed that the risks caused by air pollution are zero once the surface cover is completed.

We also needed to assess the extra risk posed by the new waste material disposed on the site, which might reach the groundwater through infiltration.

We also became aware that this risk assessment should ideally include geotechnical risk assessment, because at this site, problems involving the geotechnical aspect (slope stability) could cause the release of a huge volume of chemicals to the environment. It might also be reasonable to assess how the disposed red sludge might affect the long-term physical properties of the embankments, and thus the stability of

the whole site. These aspects were not incorporated into our health risk assessment as possible scenarios.

A deterministic risk assessment was conducted for all screened metals and other components. The air pollution scenario was based on actual dust measurements and historical monitoring data. The surface cover material had varying concentrations over three-dimensional space, thus we assumed the maximum measured concentration of the observed chemicals for our transport model. Concerning the mobilization of the investigated metals, the effect of the red sludge's high pH level had to be incorporated. We used existing column tests to assess the permeability of the red sludge "layer". Its heterogeneous composition brought rather large uncertainty to our assessment. On-site infiltration testing showed the material to have very low hydraulic conductivity in the range of 10^{-8}-10^{-9}m/s.

4. Comparison of the Two Cases

The main factor that influenced the final outcome of our risk evaluation was the fact that potential carcinogenic effects were detected in the air pollution scenario: this determined the balance of the risk components.

Another major factor that influenced the final assessment was the anthropogenic barrier of the screened toxic elements. The 7-9m thick anthropogenic layer decreased exposure levels radically so that the chance of exposure through drinking water was not significant.

The calculated risk of the disposal of the mixed hazardous waste was acceptable. We also noted that on the one hand, this "Brownfield-type" approach of using already contaminated land for the disposal of waste material is desirable. It is also true that the numerical calculations showed acceptable risk levels. However, on the other hand, disposing hazardous waste material, even at allowable concentrations of contamination, within a few dozen meters of the Danube river is not in line with our present environmental guidelines. It is acknowledged that the surface cover technology was economically viable and solved the short-term air pollution problem of the surrounding settlement, which in other ways would not have been financed.

According to our assessment, no carcinogenic risks were threatening the environment at the time, due to the surface cover technology in place. In the original case, however, carcinogenic risk was observed, mainly due to Cr^{VI} exposure through inhalation. The implementation of the surface cover was urgent and necessary to eliminate this effect, and its presence decreased the overall risk balance of the site.

Our risk assessment stated that the major risks related to the investigated sites are due to the environmental legacy of the red sludge ponds and not the surface cover technology. This risk component, however, was not the subject of the assessment. The risk assessment team recommended an extensive risk study where geotechnical and flooding scenarios are also taken into account.

5. Conclusions

Beyond the pioneer experience of the Almásfüzitő project, this case study was an interesting experience to the involved risk assessors for several reasons.

The financial (cost/benefit) aspect of the surface cover business was not available to the assessors. It is reasonable to believe, though, that the profit the site owner gained by taking over the hazardous waste from dozens of industrial players was much more than the actual cost of the surface cover implementation. The risk reduction that the site owner obtained was proved; however, it was not a breakthrough in quality of life for the people in the vicinity.

The technical aspect of this waste management leads us to think about the future of the site. The long-term fate of the red sludge is not known. One likely solution is the reuse of the rich, valuable metals contained in the sludge. In this case, the surface cover material will require new waste management solutions. Thus, its essential effect is to leave the final solution to the problem of contamination to future generations, just as the previous generation did with the original red sludge disposal.

The Almásfüzitő case was also an interesting methodical experience for demonstrating the wide scope of risk assessment applicability. Risk assessment is a unique tool that allows assessors to compare environmental threats of different nature, endpoints, and consequences on a common platform. Concentrations of contaminants are a good basis for comparison of those contaminants and their effects in the same environmental media. In this case, exposure via air pollution had to be compared to a groundwater-drinking water scenario. The risk assessment is capable of handling, comparing, and managing otherwise incomparable health threats.

Figure 6 shows that by changing the area of comparison from concentrations (as seen in traditional remediation techniques) to risk terms, a power tool placed in the hands of decision-makers. At the same time, one also has to admit that this case has also shown that some issues that need consideration during decision-making cannot be forced into the framework of any risk assessment.

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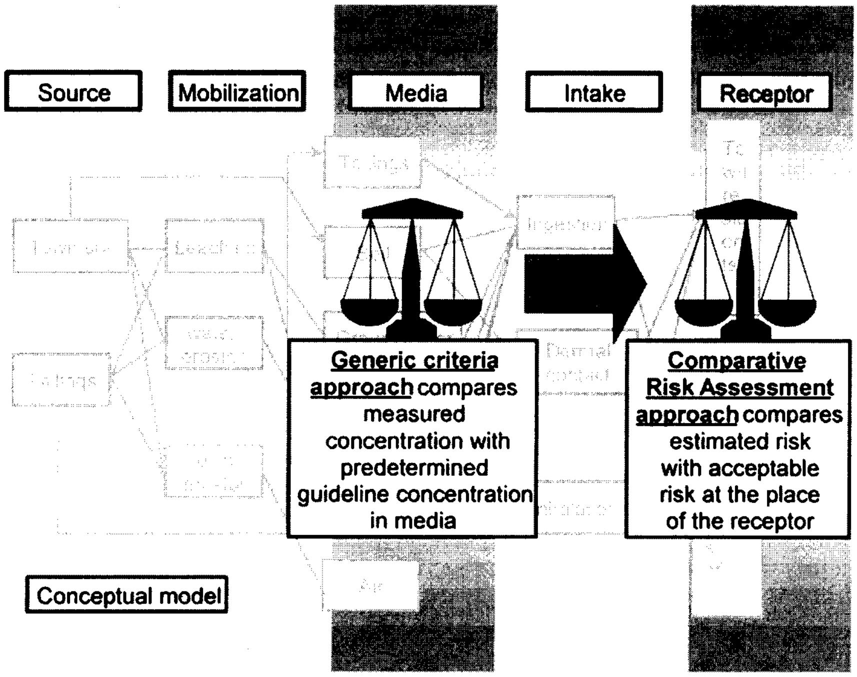

Fig 6. Risk assessment: shifting the point of comparison from the environmental media to the receptor [2]

TOWARDS A MORE COHERENT REGIONAL ENVIRONMENT AGENDA IN THE MIDDLE EAST: EXPLORING THE ROLE OF COMPARATIVE RISK ASSESSMENT

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1. The Absence of a Regional Environmental Agenda

Official publications about the environmental situation in Egypt, Jordan, Israel, and Palestine surprisingly share many of the same findings. The environment is polluted, land is degrading, biodiversity in retreat, and conditions are quickly growing worse. (Jordan Agenda 21, 2002;Israel Ministry of Environment, 2002; Palestine Environmental Quality Authority, 2002). Mobile and stationary sources of air pollution, groundwater pollution, pesticides, hazardous materials, drinking water treatment, and soil contamination are frequently measured at concentrations above European and U.S. standards and there is anecdotal evidence of high incidence of environmentally related cancers and general morbidity. At the same, no effort has ever been made in any of these countries to characterize and contrast the risks of the many environmental problems and rank them with regards to their severity.

The result is muddled and inconsistent national and regional environmental policies, where priorities are frequently decided according to the whims of politicians, organizational leaders' fundraising inclinations, media frenzy, and other irrational factors. The same is true of the sundry transboundary regional environmental initiatives that have been funded, or at least considered for funding by international donors. Mosquito control and marine parks compete with clean energy development and inter-sea canal system for funding. (El Nasser, 2003, Tal, et. al., 1995, Ecopeace, 1995). Hopes for a common ecological, or environmental health agenda are as improbable now as they were at the start of the Middle East peace process a decade ago. And yet, as the negotiations inevitably supplant the recent spate of violence, considerable resources will be made by the international community to promote some regional program, albeit an elusive one.

In driving the present development agenda in the area, each country in the area has a range of partisan interests, many of which may run counter to overall regional environmental objectives. For example, Israeli government officials have supported extensive transboundary highway projects in order to order to literally "cement" the transportation interests of Israel together with its neighbors. Jordanian support for an industrial zone contiguous to the Jordan River was assailed by environmentalists on both sides of the river. Palestinian support for international support for a Port in Gaza is considered to be detrimental to coastal preservation efforts. Even the so-called "Red-Dead Canal," a purportedly environmentally driven project to replenish the retreating

I. Linkov and A. Bakr Ramadan (eds.),
Comparative Risk Assessment and Environmental Decision Making, 125–131.

waters of the Dead Sea is increasingly challenged by the environmental community. Until now, it is not clear on what basis funding to support sustainable development in the name of confidence/peace building efforts are made. (Tal, 1997)

Scientists, policy analysts and even NGO advocates need to become aware of the merits of a systematic, analytical exercise that compares the risks associated with the region's chronic environmental deterioration. A research initiative whose ultimate goal is the clear ranking of different levels of risk for a range of environmental problems would constitute an invaluable tool for government agencies and non-government organizations, as well as the research institutes themselves in these countries. It would allow these institutions to set their own priorities and allocate the appropriate level of resources to address the most pressing environmental problems based on rigorous and objective procedures. Or, in the words of a famous comparative risk anthology, they could begin to treat – "Worst Things First." (Finkel, A. 1995). On a regional level, the results would not only provide compelling data to support Arab-Israeli cooperation directed at the most acute environmental problems, but also clearly establish a common regional agenda for action.

2. Transboundary Risk Assessment

Risk analysis is one of the most critical emerging sciences to impinge on modern environmental policies. In particular, risk-based frameworks are often developed to: integrate risk assessment and engineering options; generate performance standards; compare options for risk reduction; communicate uncertainty; and effectively allow reiteration of the decision-making process. (Konisky, 1999). Risk assessment has been used to create a common language to help reconcile competing interests in development and environmental disputes in the West. It is quite possible, that a comparative risk exercise could help shape a true common set of environmental priorities and cooperative ventures for a Middle East that largely seeks a way out of half a century of violence, but lacks a shared perspective or series of decision rules to work together.

Comparative Risk Assessment (CRA) is a methodology applied to facilitate decision-making when various activities compete for limited resources. Application of this approach is extremely flexible. CRA can be applied in very specific situations to rank risks associated with specific pollutants or environmental stressors. It is often used as a participatory process that incorporates public and stakeholder views into decision-making and promotes better understanding of environmental issues. In the present context, this participation could bring together disparate sectors within the countries of the Middle East, as well as between Israel and its neighbors.

Comparative risk assessment has become an increasingly accepted research tool and has helped to characterize environmental profiles and priorities on the regional and national level. Over thirty U.S. states (Andrews, 2002), many cities, and Native American tribes have used CRA methodology to establish environmental priorities. CRA has been used for priority setting in many industrial, developing, and transition economies. This tool is specifically recommended in the World Bank Pollution Prevention and Abatement Handbook.

In developing countries, where data are relatively sparse, comparative risk assessment has been successfully used to allocate limited resources efficiently. For example, the project Silesia (Czech Republic and Poland) involved a quantitative "screening" risk assessment to compare risks of different environmental problems in the region to develop quantitative assessments for air, surface, and ground water, food, hazardous waste, and occupational health. Economic analysis was used to develop risk management strategies and demonstration projects for the highest risks. CRA has been applied on a citywide basis in Bangkok and in Cairo to identify specific recommendations for targeted actions such as reducing lead in gasoline and managing traffic situations to decrease levels of particulate matter. Comparative risk has not, however, been implemented across national boundaries.

3. A Comparative Risk Analysis in the Middle East – Specific Objectives

There are any number of logistical, political and cultural obstacles to which the present, national focus of CRA around the world can be attributed. These include language barriers, incompatible data sets, political enmity, and diverging visions of "quality of life." Yet, from a strictly ecological, hydrological, or public perspective, a narrow, domestic, CRA effort in most of the individual countries in the Middle East, especially in the area surrounding Israel and its neighbors, will be undermined by the transboundary nature of environmental problems. It also may lead to skewed overall results from the vantage point of public health. In other words, the cumulative impact of an environmental hazard, when the populations of three or four nations are totaled together may warrant higher ranking than it would based on the exposed population a given country. Moreover, while a domestic CRA effort may be able to contrast the relative severity of different health risks, risk management strategies that depend on interventions from neighboring countries will not be considered. It is also uncertain whether the data are available to enable an individual country to fully characterize its risk portfolio. Certainly, Palestinian environmental researchers have often claimed that their efforts are stymied by data which are either only available in Hebrew – or not made available at all.

A risk assessment that considers issues associated with river contamination and restoration is instructive. Within Israel and the Palestinian Authority there are fifteen streams that cross the Palestinian/Israeli border. Twelve of these are major streams that flow year-round in a westward direction toward the Mediterranean Sea. All of them originate in watersheds located in the Palestinian Authority, or in lands that will eventually be outside Israeli jurisdiction, and then flow into Israel. Similarly, there are three major streams with easterly flow to the Dead Sea or Jordan River that originate in Israel and cross into the Palestinian Authority. (Tal, 2002) At least part of each of these streams can be defined as highly polluted, posing a health hazard to users, endangering flora and fauna and leaving them unfit for recreational or consumptive uses. Their restoration should be considered in a multi-lateral context.

There are several other media in which risk is driven by multi-national factors. Israel's groundwater increasingly reflects the sewage profile of the West Bank (Avisar, 1996) while Jordanian air quality is often driven by the mobile source emissions of Tel

Aviv and Jerusalem (Tal, 1996), to the West. Pesticide usage affects the mutations and immunity of different insects who move across national borders. (Isaac, 1995) Clearly, these issues need to be considered within the entire water or air shed in which they are created and in which they need to be treated.

What then should be the objectives of a regional comparative risk assessment in the "peace region" of the Middle East involving transboundary pollution in Jordan, Israel, Palestine and Egypt (as well as Lebanon and Syria)? There are several which are particularly prominent:

1. To collect a wide variety of data regarding the critical environmental exposures in participating countries;
2. To identify the gaps in existing knowledge that need to be filled in to fully evaluate environmental and ecological risks and conduct preliminary monitoring to fill them;
3. To characterize the associated environmental risks in each of these countries with regard to impact on human health, ecological conditions and socio-economic factors;
4. To compare the relative risks within each country in order to better establish coherent public policy priorities; and
5. To compare the risks existing in each of the countries with those associated with transboundary environmental exposures to help define a coherent regional environmental agenda and rank the relative severity of a variety of environmental problems.
6. To design a "risk management" strategy to reduce these risks with an eye towards directing international investment which might be funded by the donor community towards projects which produce optimal risk reduction results.

Even though comparative risk assessment is widely used, the risk ranking methodology has not been standardized. A regional study, however, need not "reinvent the wheel" and can rely heavily on the comparative risk protocols developed in the recently completed New Jersey Comparative Risk Project or on recommendations and recent experience reported by the World Bank (1998) and EPA (1993). Expert committees can approve a variety of decision analytic techniques based on the recommendations of the individual experts or project managers that would represent each country.

4. Methodology and Project Management

A CRA project generally has two stages: risk comparison and ranking; and strategic analysis and priority setting. For the first stage, a transboundary comparative risk project in the Middle East should start with an committee of experts from a variety of disciplines that can generate a list of those environmental issues that are regional in context and worthy of characterization within a transaboundary analysis. The committees should be composed of diverse representatives of environmentalist, business, academic, civic, and other perspectives *from each participating country* to produce a list of key environmental issues and specify how to assess these issues'

impacts on health, ecological quality and socio-economic conditions, ultimately ranking their relative severity.

Deciding which environmental problems are in fact "regional" and sufficiently severe to warrant analysis will not be a trivial task. In developing these lists, the criteria and issues for addressing health and ecological risks that were used by various U.S. states and cities as well as other countries will be instructive. (U.S. EPA, 1993) In cases of disagreement, a default arrangement might be based on procedures utilized under the New Jersey comparative risk project that are fully documented on their web site including expert selection, participation criteria, etc. It may be valuable to incorporate an external expert from outside the region with considerable practical experience in conducting comparative risk assessment who can serve an arbitrator and guide in the process.

The committee would then approve local experts to form three Expert Working Groups (one each on health, ecological quality, and socio-economic conditions) whose ultimate job will be the estimation of environmental issues' relative impacts, using the best available science consistent with a broad and timely analysis. Each of the Working Groups should be chaired by a leading scientist in the field or run through a "rotating" chair. In either case, the groups must be supported by full-time graduate assistants from each country whose task would be to administer the program and ensure expeditious information sharing. The Expert Working Groups would be responsible for implementing the data collection as well as the supplementary monitoring.

Once a regional issue list is compiled, technical work groups can start to collect and analyze the best data available and describe the level of risk for each issue. Here, teams should have representatives from each country, if for no other reason to facilitate data collection from local agencies and research institutions.

A general risk assessment framework should be used in data analyses. The stressors or sources of risks need to be grouped to facilitate the analysis. In the human health risk assessment, the estimates can be made of the population exposed to each stressor and the concentration of the particular stressor and exposure pathways. Here again, it is likely that the populations at risk from a particular exposure will include residents of more than one nation. For example, the sewage of Kalikiliya constitutes a risk to Palestinians and Israelis alike. Particulates from coal-fired power plants in Israel reach nationals of three different countries at least in measurable concentrations.

As is standard for CRA studies, the risks to typical exposed individuals can be estimated, using either cancer potency factors for carcinogens or a reference dose for non-carcinogens. Population-level risks will be assessed by multiplying the individual risks by the number of people assumed to be similarly exposed, regardless of their actual geo-political affiliation. Special population groups could be considered in each and across each country.

If the study went beyond human health to include ecological risk assessment, risk to specific ecological receptor communities should be evaluated. Weight-of-the-evidence approach can be used to assess these risks. At the next stage, stakeholder values will need to be used to compare and rank different ecosystems and communities. This may constitute the most challenging part of the study, inasmuch as cultural perceptions as well as actual exposure levels may differ widely between nations. As a

final step, the technical groups, together with representatives of stakeholders and the public from each country should compare and rank the risks on the basis of criteria specific to each risk category (health, ecosystems, or quality of life) with the international advisory committee integrating all the issues into a single ranked list. Web-sites with English, Arabic and Hebrew sections would be valuable in disseminating the results of the study and reporting on its progress.

5 Innovation and Contribution to Regional Cooperation

There are several types of benefits which a CRA in the Middle East could generate. To begin with, a regional CRA will produce a baseline comparative risk assessments in the Middle East and for the first time allow scientists and decision makers to consider the full-realm of their common environmental challenges based on empirical and systematic analysis. Such a project would also constitute the first-ever, comprehensive, trans-boundary comparative risk assessment internationally. The resulting protocol and management models, evaluating a multiplicity of multi-lateral environmental problems and ranking them, not withstanding the diverse cultural perspectives and national interests could be utilized by other countries who also face a range of transboundary environmental challenges.

The potential contribution of such a scientific initiative to the fragile fabric of cooperation in this volatile region should also not be underestimated. A CRA project would offer a unique opportunity for scientists in the "peace region" to convene in an apolitical framework and undertake a comprehensive evaluation of their common problems with the goal of forging a shared environmental regulatory and research agenda. The Egyptian, Palestinian, Jordanian, and Israeli researchers and graduate students who would ultimately work together on the project could continue to serve as a cadre of risk assessors who would help their own countries do a better job of formulating a rational public health and environmental agenda.

Decision makers in the area of environmental management would at long last receive a systematic, unbiased analysis of the relative severity of environmental problems in their countries. Risk analysis as a new and up-to-date, unfamiliar scientific discipline will be introduced to Egypt, Jordan, Israel, and Palestine. A regional agenda for common action will be formulated based on the severity of the problems, rather than political exigencies. This will be of particular interest to any donor countries who are interested in receiving maximum return on their investment on sustainable economic initiatives that strengthen the peace process or assisting the countries work together on common environmental challenges. Furthermore, public interest NGOs that are active in these countries will have access to information critical to the conducting of public awareness campaigns for a range of environmental media. Finally, the populations in these four countries will have a better sense of the risk and where societal resources should be directed to improve their common environment.

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LESSONS FROM THE NEW JERSEY COMPARATIVE RISK PROJECT

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Abstract

Experience is a great teacher, and learning from someone else's experience can make the lessons much less painful. This chapter shares lessons learned from the recently completed New Jersey Comparative Risk Project (NJCRP). It briefly describes the project and offers a preliminary evaluation of its adequacy, value, effectiveness, and legitimacy. The main purpose of this broadly scoped project was to inform a state regulatory agency's strategic decisions. The project involved a large number of technical experts from a variety of fields, plus public officials, high-profile stakeholders, and members of the general public over a four-year period. It gathered and organized a vast amount of useful information, but found that there was still an inadequate scientific basis for a precise single ranking of environmental threats. The NJCRP instead developed policy findings using a humbler approach that involved less aggregation. Highlighted environmental threats for New Jersey included land use change, indoor environmental problems, a set of traditionally regulated pollutants, and invasive exotic species.

1. New Jersey Comparative Risk Project

The Commissioner of the New Jersey Department of Environmental Protection (NJDEP) launched this strategic planning activity in 1998, after agency managers had advocated for it over the preceding decade [1]. Partial funding came from the U.S. Environmental Protection Agency (USEPA), which in 1987 launched the comparative risk paradigm with its *Unfinished Business* study [2]. Designers of the New Jersey project benefited from the collective lessons learned over a decade of experience involving more than twenty comparative risk projects carried out by U.S. states and territories [3, 4].

The NJCRP employed an analytic-deliberative process of the type endorsed by the National Research Council in its rewrite of the famous "red book" on risk assessment [5, 6]. It was a broadly scoped project that evaluated the relative human health, ecological, and socioeconomic impacts of 88 environmental threats. It had a statewide scope, but it also took care to identify localities and populations facing elevated risks. The project adopted a relatively short time horizon, and it considered

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only the impacts likely to occur within the next five years. It relied on a separate trend variable to track slowly emerging threats like global climate change.

The NJDEP Commissioner charged the NJCRP with four major objectives. First, it should compare New Jersey's environmental problems to one another in a systematic way, ideally in the form of a ranked list. Second, it should identify knowledge gaps. Third, it should improve the agency's basis for evaluating alternative environmental management strategies. Finally, it should promote discussion within New Jersey on the state's environmental challenges.

The final report of the NJCRP came out in 2003. The technical work had been finished over a year previously, but in the interim a new Governor from a different party had taken office, and the new administration needed to take ownership of the effort. Equally important, that year was also needed for a substantial effort to write a clear, persuasive report and produce a user-friendly website for the project.

The remainder of this chapter mines the New Jersey case for useful lessons. This case is worth reviewing because its designers consciously innovated, but not necessarily because they did everything right. No real-life project does that. Arguably, this project took too long, depended too heavily on volunteer labor, never satisfied all stakeholder perspectives, and glossed over inconsistencies. But it did successfully reach completion, and it offers several useful lessons.

2. Evaluation Criteria

Good scientific policy analysis should be adequate, valuable, effective, and legitimate [7]. These four criteria offer a basis for evaluating the inputs, processes, and outputs of the NJCRP, once defined.

- *Adequacy* is a measure of how authoritative the science is. Are the data, models, and conclusions valid and reliable?
- *Value* measures the incremental contribution from several perspectives. Internally within the analytical community, did the project innovate? Externally for the public policy community, did the project reveal important insights? Personally, for those involved, was it worth the effort?
- *Effectiveness* measures the visible impact of the project. Did anyone notice it?
- *Legitimacy* measures the extent to which the project is viewed as being desirable, proper, and appropriate [8]. Do people accept that it is authoritative and in the public interest?

The criteria are interlinked, of course. Scientific adequacy contributes to legitimacy, which when coupled with value, can improve the overall effectiveness of analysis in influencing public policy. Yet they measure distinct aspects of a project.

3. Definition of Risk Used

Unlike highly detailed technical risk assessments supporting regulatory decisions, the risk analysis approach used in the NJCRP is more broadly scoped, adapted to support the agency's strategic planning efforts. How do the two associated risk definitions differ?

Most technical risk assessments strive to develop a highly defensible single number. In such assessments, risk is an expected value. Risk is the likelihood an adverse outcome occurs, equaling a probability times an outcome. For example, the approximate average annual risk of death by lightning in the United States = (1 chance in 4 million) x (death) [9]. Comparative risk in such a context is a process of direct comparison on a common scale, commonly displayed as a risk "ladder" ranging from high to low risks. For example, [10] claims that the regulatory system is flawed because a risk ladder shows an orders-of-magnitude difference in average annual death rates due to unregulated smoking and regulated asbestos exposure.

In a strategic planning context, single-dimensional risk ladder comparisons have only limited value. That is because risk comparison is really a multidimensional task. Policymakers and the public not only care about the human health endpoint of death, but also other human health impacts, ecosystem impacts, and socioeconomic impacts. It also matters to public policy whether the threat is local or widespread, voluntary or involuntary, and whether the impacts are chronic or acute [11]. Population-average expected values can also be misleading when the incidence of a risk varies systematically across elements of a population. Finally, the incertitude associated with a risk estimate is itself relevant to making public policy.

Reflecting the multi-dimensional richness of risk comparisons is clearly a daunting analytical task. Compounding the challenge is the inevitability of resource constraints. It is not unusual for regulators and regulated parties to spend millions of dollars on detailed risk analysis leading to a proposed regulation. Doing so for dozens of threats in a state-level strategic planning context would be cost-prohibitive. Thus, comparative risk analysis for strategic planning purposes must consider a broad range of impacts on a shoestring budget. That is a recipe for scientific inadequacy, and it can be expected to weaken the legitimacy of the findings.

4. The Analytic-Deliberative Process

Weber suggests that there are parallel routes to legitimacy: the authoritative "numinous" legitimacy accorded god-kings and respected scientists, and the democratic "civil" legitimacy accorded to fair and transparent political processes [12]. In a similar vein, Simon distinguishes between substantive and procedural rationality, implying that defensible decisions are rational in both senses of the word [13]. Williams and Matheny add a third route to legitimacy by distinguishing among scientific authoritativeness (a managerial conception), officially sanctioned processes (a pluralistic conception), and open community access (a communitarian conception) [14]. Andrews argues pragmatically that legitimacy has four complementary sources: involvement of experts, public officials, professional stakeholders, and the general public [3].

Previous experience suggests that for strategic planning applications of comparative risk analysis, there is built-in scientific inadequacy. This means that expert involvement on its own is particularly unlikely to persuade people that a study's findings are legitimate. Hence procedural considerations gain in importance.

The designers of the NJCRP chose to involve experts in Technical Work Groups (TWGs), stakeholders and public officials on a Steering Committee (SC), and members of the general public through focus groups, surveys, newsletters, open meetings, and a website. Thus the project entrained all four types of potential contributors to legitimacy identified previously. The SC was in charge, soliciting and receiving expert reports from the TWGs and lay input from members of the general public, using a process that was carefully structured and highly interactive.

TABLE 1: Environmental Threats Assessed in the NJCRP (figure threat number key)

1,3 Butadiene (1)	Green/red tides (33)	Pesticides-outdoor (56)
Acid precipitation (2)	Habitat fragmentation (67)	Pesticides-water (56)
Acrolein (3)	Habitat loss (67)	Petroleum spills (57)
Airborne pathogens (4)	Hantavirus (32)	Pets as predators (58)
Arsenic (6)	Hemlock woolly adelgid (34)	Pfiesteria (59)
Asian longhorn beetle (8)	Impervious surface increase (67)	Phosphorus (60)
Benzene (9)	Inadvertent animal mortality (36)	Phthalates (35)
Blue-green algae (33)	Indoor asthma inducers (37)	Polychlorinated biphenyls (PCBs) (61)
Brown tide (10)	Indoor microbial air pollution (37)	Polycyclic aromatic hydrocarbons (PAH) (62)
Cadmium (12)	Invasive plants (38)	QPX in shellfish (63)
Carbon Monoxide (13)	Land use change (67)	Radionuclides (NA)
Catastrophic Radioactive Release (NA)	Lead (39)	Radium (64)
Channelization (16)	Legionella (40)	Radon (64)
Chromium (17)	Light pollution (41)	Road salt (65)
Copper (19)	Lyme disease (21)	Secondhand Tobacco Smoke (27)
Cryptosporidium (20)	Mercury (44)	Starlings (66)
Deer (21)	Methyl tertiary butyl ether (46)	Sulfur oxides (68)
Dermo in Oysters (22)	MSX in oysters (22)	Thermal pollution (70)
Dioxins and Furans (23)	Nickel (47)	Tin (71)
Disinfectant by-products (24)	Nitrogen oxides (49)	Ultraviolet radiation (72)
Dredging (25)	Nitrogen pollution (48)	Volatile Organic Compounds (VOC) (73)
EHD in Deer (26)	Noise pollution (50)	VOC-carcinogenic (73)
Endocrine disrupters (35)	Off road vehicles (52)	VOC-non-carcinogenic (73)
Extremely low freq./magnetic radiation (51)	Overharvesting (marine) (53)	Water overuse (74)
Floatables (28)	Ozone (ground) (54)	Waterborne pathogens (75)
Formaldehyde (29)	Particulate matter (55)	West Nile virus (76)
Geese (30)	Pesticides (56)	Zebra mussels (77)
Genetically modified organisms (31)	Pesticides-current (56)	Zinc (78)
Greenhouse Gases (18)	Pesticides-food (56)	
	Pesticides-historic (56)	
	Pesticides-indoor (56)	

The SC scoped the project very generally and then finalized details in negotiation with the TWGs and with guidance from public input. For example, the SC directed the TWGs to study eleven broad categories of environmental threats: metals, other inorganic chemicals, organic chemicals, radiation, plants and animals, microorganisms, indoor air quality, thermal/light/noise, physical transformations of

land and water, climate change, and natural resources use and impacts. The TWGs came back to the SC with hundreds of stressors within these categories. Public input by means of focus groups, booths at public events, and newsletter tear-offs also added a few items to the list. Ultimately, 88 threats were evaluated (see Table 1).

The experts involved in the NJCRP were organized into three TWGs. A Human Health TWG included approximately twenty toxicologists, epidemiologists, physicians, and other health scientists. They evaluated multiple impacts, including cancer and noncancer mortality and morbidity. An Ecological TWG included some 25 ecologists and biologists. They evaluated impacts to ecosystem health by ecosystem type and by watershed. A Socioeconomic TWG included about fifteen economists, planners, sociologists, and other social scientists. They evaluated impacts on property values, employment, out-of-pocket costs, aesthetics, and worry. Table 2 describes the impact assessment calculus.

TABLE 2: Impacts Evaluated in the NJCRP

Socioeconomic impacts were assessed for five endpoints: Property Values, Employment, Direct Costs, Aesthetics, Psychological Well-Being Rating system: Each type of impact receives a rating and a level of confidence (low, medium, high) in that rating. Impact ratings are the product of 3 elements (severity x irreversibility/duration x extent). Range of ratings per endpoint is 1 to 27. Severity, irreversibility/duration, spatial extent, level of confidence, and catastrophic potential are all rated on a Low (1), Medium (2), High (3) scale. Cut-point between Medium and High is set so that a High rating implies major socioeconomic significance, e.g., for employment the severity cut-point is the amount of job losses occurring during the early-1990s economic recession. Cut-point between Low and Medium is one order of magnitude lower. Overall rating is the sum of the five different impact type ratings. Range of overall ratings is 5 to 135. Also assessed: Hot Spots/Populations at Elevated Risk, Catastrophic Potential, Trend (rated on –3 to +3 scale of deterioration or improvement)
Ecological impacts were assessed for five ecosystem types: Inland Waters, Marine Waters, Wetlands, Forests, Grasslands Rating system: Ecosystem impact is calculated as the product of 3 elements (severity/irreversibility x temporal frequency x spatial magnitude). Each element is rated on a Low (1), Medium-Low (2), Medium (3), Medium-High (4), High (5) scale. Range of ratings per ecosystem type is 1 to 125. Cut-points are established by repeatedly calibrating ratings against the consensus of TWG members. Overall rating is the average of the five different ecosystem ratings. Range is 1 to 125. Also assessed: Hot Spots/Populations at Elevated Risk, Catastrophic Potential, Trend
Human Health impacts were assessed across the aggregate of health endpoints, not standardized. Rating system: Health impact is calculated as the product of 3 elements (severity x size of affected population x vulnerability). Each element is rated on a Low (1), Medium-Low (2), Medium (3), Medium-High (4), High (5) scale. Cut-points are established by repeatedly calibrating ratings against the consensus of TWG members. Range of overall ratings is 1 to 125. Also assessed: Hot Spots/Populations at Elevated Risk, Catastrophic Potential, Trend

The TWGs produced a vast amount of information, including 178 impact assessments (88 environmental threats x 3 major impact categories minus unnecessary ones), most of which exceeded 10 dense pages each. Often the authors provided

quantitative estimates of impacts, but in many cases a qualitative estimate was all that was available. Each assessment was internally peer reviewed, and each ecological and socioeconomic assessment was also externally peer reviewed. Each TWG separately developed a scoring system to rank the many environmental threats overall and along dimensions such as severity, irreversibility, and geographic extent for specific impact categories. The TWGs also characterized the trend (improving or deteriorating?), catastrophic potential, and level of expert understanding associated with each environmental threat, although these dimensions were not included in the aggregate scoring system. Thus the TWGs delivered much interesting and useful analysis to the SC and overwhelmed them in the process.

Although the SC had requested quantitative impact assessments and precise rankings, the experts on the TWGs instead delivered quasi-objective/subjective, multi-dimensional scores backed by assessments of widely varying depth and quantitativeness. That was the best they could do given available scientific evidence and staff resources. The SC then sought a way to transform this mass of incomparable information into a concise overall ranking of New Jersey's environmental problems. Such aggregation challenges often occur in environmental decision-making [15]. The SC tried four alternative approaches, described in Table 3.

TABLE 3: Aggregation Techniques Tested in the NJCRP

Aggregation Technique
Alternative #1: Holistic Ranking SC members discuss and holistically create a ranking. Informal analytics, but potentially a legitimate process because SC members are officially appointed and "representative."
Alternative #2: Constructed Ranking [16] Design a rating & weighting scheme for ranking. TWGs develop expert ratings for various impacts of environmental threats as described in Table 2. SC members and the public specify subjective weights for various categories of impacts (human health, ecological, etc.). Apply weights to ratings, calculate total scores, rank threats based on weighted scores as shown in Table 4. Calculate confidence intervals around scores.
Alternative #3: Triangulation (Based on [17]) SC members each develop a holistic ranking of threats. SC members each specify weights and create a constructed ranking (based on rating & weighting calculation from #2). SC members individually triangulate between 2 approaches. SC as a whole creates an overall ranking (by voting or by consensus).
Alternative #4: Sorting (Based on [18]) Design a rating-only scheme for ranking, no weights. TWGs develop expert numerical ratings for various impacts of environmental threats as described in Table 2. TWGs establish cut-points distinguishing categorical Low, Medium-Low, Medium, Medium-High, and High overall ratings for each impact category, using natural breakpoints & expert consensus. Sort into a categorical ranking of threats for each impact category. Remove the threats rated Low in all impact categories. Focus SC's attention on threats with High ratings in several impact categories. Identify patterns and clusters among various multi-criteria rankings. Highlight these in final report.

5. Analytical Issues

Rating-and-weighting is a time-honored approach to aggregation. Table 4 shows an illustrative simplified calculation performed for the NJCRP to support the Constructed, Triangulation, and Sorting approaches to the aggregation problem. Given ratings (by impact category) and weightings (based on public opinion surveys or other preference data), analysts could develop a precise score for each environmental threat. Following experimentation with a variety of normalization and transformation schemes, analysts determined that the specific conversion from a five-point scale to percentage weights shown was reliable and representative, while still delivering substantial variance.

TABLE 4: Illustrative Constructed Risk Ranking Calculation (Deterministic)

Major Impact Class	(a) Un-weighted TWG rating	(b) Weight from survey	(c) Transform (=5-b)	(d) Invert (=1/c)	(e) Normalize (=d/Σd)		Weighted Rating (=a*e)
Human Health (Range = 1 to 125)	2	4.8	0.2	4.9975	0.644		1.29
Ecological Quality (Range = 1 to 125)	80	4.5	0.5	1.9996	0.257		20.60
Socioeconomic (Range = 1 to 135)	60	3.7	1.3	0.7692	0.099		5.946
		(Range = 1 to 5)		7.7663	1.000		Total
Environmental Threat ID#: **67**		Source of Weights: NJ Public				Aggregate Score (Range = 1 to 300)	**27.83**

TABLE 5: Scientific Uncertainty Levels Used by the Socioeconomic TWG

Low Uncertainty	Medium Uncertainty	High Uncertainty
Impact estimate is quantitative and well documented. Scientific consensus exists on estimating approach. New Jersey-specific estimate used. It is highly probable (68% or better, i.e., one standard deviation) that the reported score is correct.	Some documentation exists. A literature relying on this estimating approach exists. Some NJ-specific data were used. Confident that, if scores above are wrong, they are, on balance, only off by one (High vs. Medium). There is at least a 50% probability (even odds or better) that the reported score is correct.	Impact estimate is qualitative and poorly documented. No scientific consensus exists on estimating approach. No NJ-specific estimate available. Scores above are, on balance, quite arbitrary, and could be off by more than one (High vs. Low). It is no more probable that the reported score is correct than that a lower or higher score is correct, so the probability that the reported score is correct is about 34%.

An important innovation in the NJCRP was to reflect incertitude about the scientific facts and public values upon which the analysis was based. Using a Monte Carlo analysis module available as an add-in for a spreadsheet, NJCRP analysts drove the deterministic calculations in Table 4 to simulate stochastic results. Decisioneering's

Crystal Ball add-in simulated stochastic outcomes by running thousands of deterministic simulations with variable values sampled from distributions input for the stochastic ratings and weights.

The TWGs specified a level of confidence (Low, Medium, High) for each impact assessment performed. Table 5 provides illustrative definitions. Analysts then constructed stochastic ratings representing the project's uncertain "facts" from these inputs.

Note that Table 5 shows the definitions used by the Socioeconomic TWG as applied to impact ratings scored on a 3-point Low, Medium, High basis, whereas the other TWGs and all applications to 5-point and greater scales relied on less formal definitions.

Stochastic weights representing New Jersey's uncertain "values" were constructed in a similar manner, as follows. Members of the public (not a random sample, but weights seemed reliable) filled out a survey eliciting the relative weights they assigned to various human health, ecological, and socioeconomic impacts (see Table 6). The distribution of weights across the survey sample (see Figure 4) was used in place of single-number weights to drive the Monte Carlo simulations. To help individual SC members aggregate while acknowledging incertitude using the Constructed and Triangulation approaches, a personal questionnaire elicited a confidence level for each set of weights assigned (see Table 6). Analysts tested a variety of distributions and determined that the specific conversions used were reliable and representative.

TABLE 6: Eliciting Uncertainty Levels for Values (Assigning Weights)

When you think about the relative severity of various environmental problems, how much importance do you attach to each of these three major types of impacts? (Circle one per row)

	Unimportant			Very Important	
Human health impacts (human discomfort, illness, disability, or death)	1	2	3	4	5
Ecological impacts (loss of biodiversity, ecosystem function, habitat, or species)	1	2	3	4	5
Socioeconomic impacts (property values, employment, costs incurred, aesthetics, psychological effects)	1	2	3	4	5

Rate the confidence--the degree of certainty--you have in your answers above.

	Low--very unsure			High--quite certain	
	1	2	3	4	5

Figures 1, 2, and 3 display the NJCRP's uncertain "facts," showing stochastic rankings of environmental threats based only on Human Health, Ecological, and Socioeconomic TWG inputs, respectively. Figure 4 displays the NJCRP's uncertain "values," showing the distribution of weights elicited in the public opinion survey. Figure 5 combines both "facts" and "values" into an overall stochastic ranking. Shown in the figures are the means and first standard deviations of the distributions of scores, which serve as the functional equivalent of confidence intervals in the interpretation of this Monte Carlo analysis.

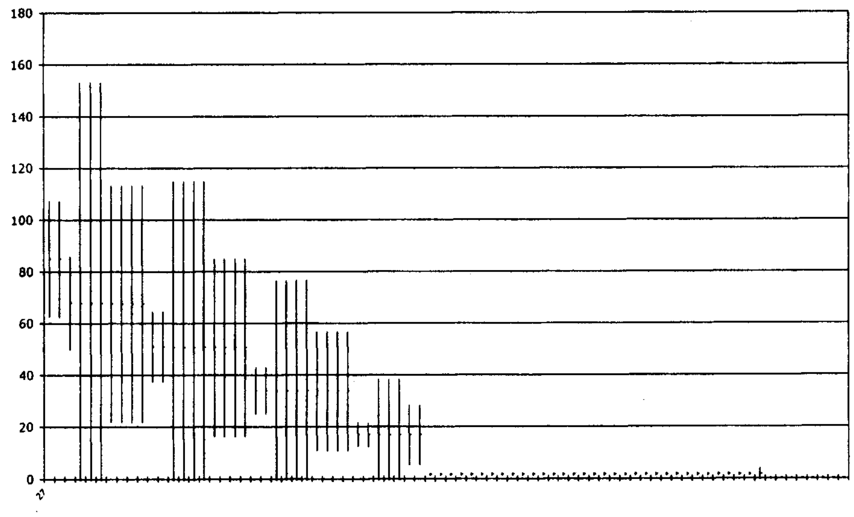

Fig. 1. Human Health Impact Ranking

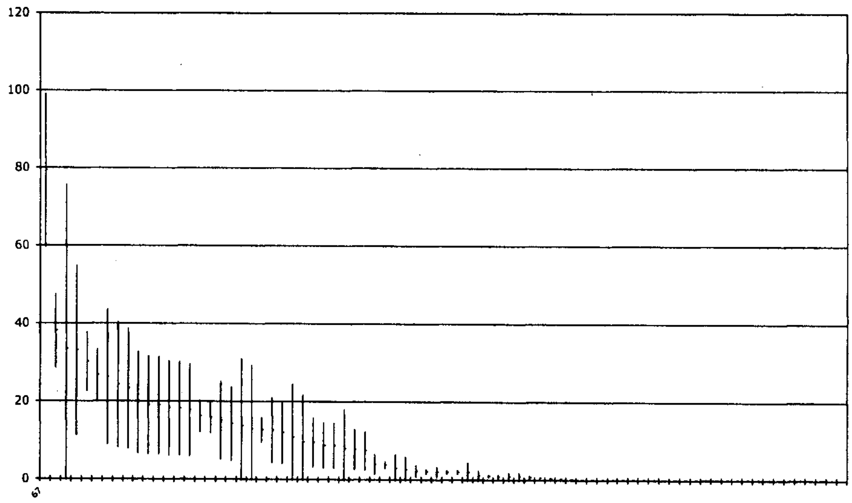

Fig. 2. Ecological Impact Ranking

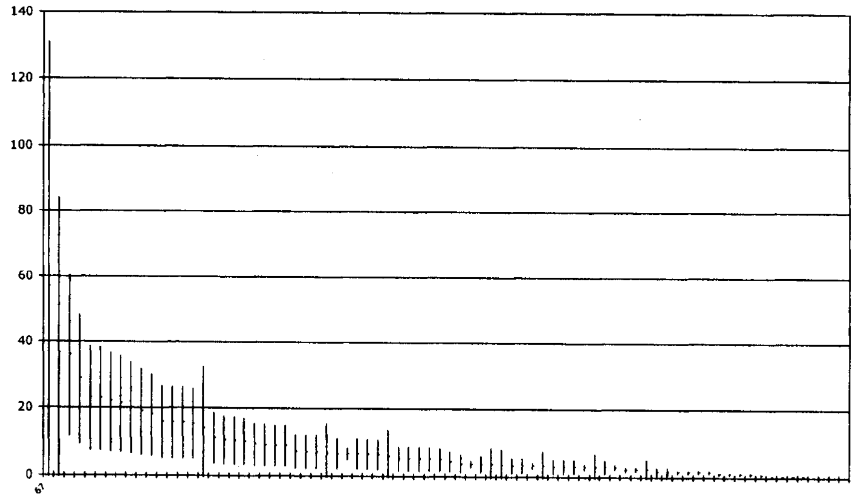

Fig. 3. Socioeconomic Impact Ranking

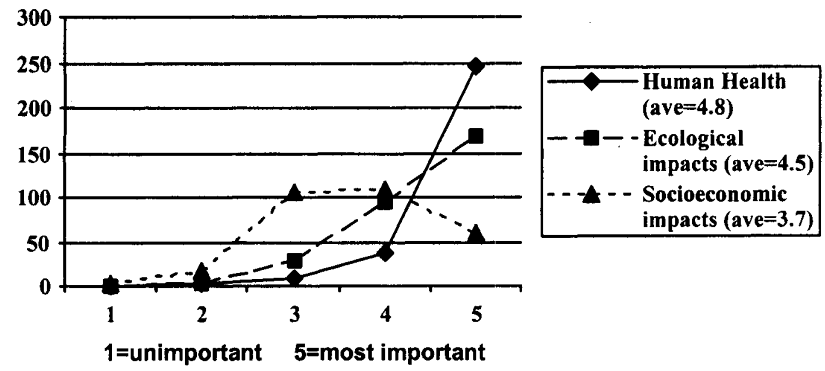

Fig. 4. Distributions of Weights Based on Public Opinion Survey (N = 300)

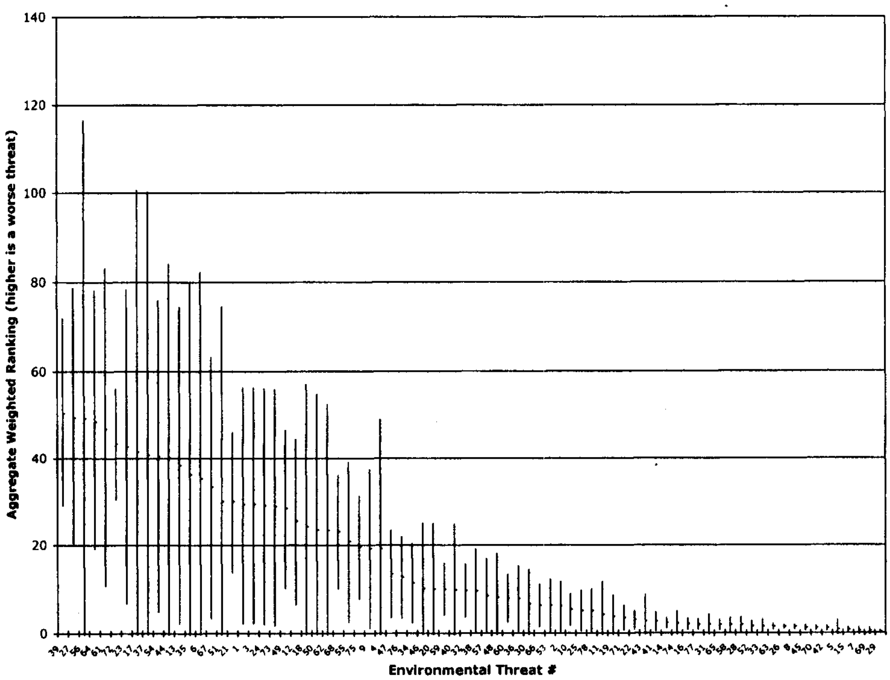

Fig. 5. Weighted Ranking

The simple rating-and-weighting utility function employed here makes strong assumptions about monotonicity, independence, and transitivity that warn us to be cautious when interpreting the results [16]. SC members showed much sensitivity to

this issue. They were very uncomfortable applying fixed weights across threats, because they could identify cases where a calculation that always gave more weight to health over ecological over socioeconomic concerns represented bad public policy.

The most important policy conclusion, however, is apparent in the overlapping score ranges displayed in Figure 5. Given that both the factual and value-based components of the ranking process exhibited a substantial level of stochastic uncertainty, the final aggregate ranking was so imprecise as to be meaningless. And thus far, other types of incertitude—structural uncertainty and scientific ignorance—were not even addressed [19]. The SC therefore adopted a humbler Sorting approach to the aggregation problem.

6. Sorting Out the Policy Lessons

If an environmental threat has high human health, ecological, and socioeconomic impacts, then it is probably more important than one with low impacts across the board. The "dominance" decision rule, which has a stochastic counterpart, is widely accepted and easy to understand [18]. Simple sorting techniques aided the NJCRP analysis even though incertitude was pervasive.

Sorting revealed that there were many environmental threats that had low impacts on all measured impact categories, allowing the SC to focus its efforts on understanding the relatively few remaining threats (see Table 7, which excludes lower-ranked threats). Among these, there were two clusters: some threats had high human health and socioeconomic impacts, and another set had high ecological and socioeconomic impacts. No threat exhibited both high ecological and human health impacts.

TABLE 7: Environmental Threats Ranked High or Medium-High (cluster key)

Human Health Impacts	Ecological Impacts	Socioeconomic Impacts
Carbon Monoxide-indoor (II)	Habitat fragmentation (I)	Arsenic (III)
Dioxins/Furans (III)	Habitat loss (I)	Deer (I)
Indoor Asthma Inducers (II)	Hemlock woolly adelgid (IV)	Secondhand tobacco smoke (II)
Lead (II)	Increase in impervious surface (I)	Indoor asthma inducers (II)
Ozone (ground level) (III)	Mercury (III)	Land use change (I)
Particulate matter (III)	Pesticides-historical (III)	Lead (II)
Pesticides-indoor (II)	Ultraviolet radiation (III)	Particulate matter (III)
Polychlorinated biphenyls (PCBs) (III)		PCBs (III)
Radium (III)		Pesticides (III)
Radon (II)		Petroleum spills (III)
Secondhand tobacco smoke (II)		Phosphorus (III)
VOC-Carcinogenic (III)		Ultraviolet radiation (III)

Sorting helped the SC identify four clusters of highly ranked threats that deserved increased attention from policymakers. Cluster I included elements of land use change (suburbanization, brownfields abandonment, habitat fragmentation, and increased impervious surface cover, among others) which had high ecological and

socioeconomic impacts. Cluster II included a variety of indoor environmental threats (secondhand tobacco smoke, carbon monoxide, radon, asthma inducers, lead, and pesticides, among others) which had high human health and socioeconomic impacts. So did Cluster III, which consisted of traditional pollutants already being regulated (outdoor pesticides, ozone, particulates, nitrogen oxides, and a few others). Finally, Cluster IV included invasive exotic species (plants such as phragmites, and animals such as hemlock wooly adelgids) which had medium-high or high ecological impacts.

TABLE 8: Environmental Threats with Highly Uncertain Impact Assessments

	High Incertitude	Medium-High Incertitude
High/Medium-High Impacts	Indoor Asthma Inducers (H) Land Use Change (S) Lead (S) Pesticides-historical (E) Pesticides-indoor (H)	Arsenic (S) Deer (S) Indoor Asthma Inducers (S) Particulate Matter (S) PCBs (S) Pesticides (S) Petroleum Spills (S) Phosphorus (S) Radium (H) Secondhand Tobacco Smoke (S) Ultraviolet Radiation (S)
Medium Impacts	Chromium (H) Endocrine Disruptors(H) Indoor Microbial Air Pollution (S) Legionella (H) Mercury (H) Pesticides-food, outdoor, water (H) Phthalates (E)	1,3-Butadiene (H) Acrolein (H) Endocrine Disruptors (E)
Low/Medium-Low Impacts	Airborne Pathogens (H) Arsenic (E) Copper (E) Cryptosporidium (H) ELF/EMF (H) Genetically Modified Organisms (E) Greenhouse Gases (H, E) Indoor Microbial Air Pollution (H) Light Pollution (E) MTBE (H) Nickel (E) Noise (H, E) PAHs (H) Pesticides-current (E) Pets as Predators (E) QPX Parasite in Shellfish (E) Road Salt (E)	Chromium (E) PAHs (E) ELF/EMF (E) Waterborne Pathogens-drinking water (H) West Nile Virus (H)

Note that E = Ecological impacts, H = Human health impacts, S = Socioeconomic impacts

Table 8 shows the environmental threats sorted on the basis of their impacts' associated scientific uncertainty (low- and medium-incertitude cases are excluded to save space). It shows that the major findings above are fairly robust, with a few

exceptions, including inadequate human health impact data on indoor asthma inducers, and inadequate ecological impact data on historical uses of pesticides. It also identifies several important knowledge gaps, including a general lack of transferable data on socioeconomic impacts, inadequate ecological inventories, and missing exposure information for human health impacts at the state and local level of analysis.

7. Evaluating the NJCRP

How did the NJCRP fare in this preliminary evaluation?

Adequacy: The analysis was inadequate to perform the ranking task that had been requested by the project's sponsor. There was neither an adequate base in general scientific knowledge, nor adequate analytical resources available to gather relevant information. This is almost always the case with broadly scoped comparative risk projects. However, the analysis was adequate to reveal a set of important policy conclusions.

Value: The NJCRP provided novel insights about the seriousness of indoor environmental problems that were valuable to external constituents. It innovated in its treatment of incertitude, providing value to internal constituents. Value to project personnel varied widely, providing useful networking and learning opportunities but also eating up huge amounts of time.

Effectiveness: The NJCRP has enjoyed positive press coverage, and its ability to survive a gubernatorial transition surprised many. The findings on land use change have been embraced by the new governor for their support of his "smart growth" agenda. The findings on indoor environmental threats have spurred new interagency cooperation between the state departments of health and environmental protection. Whether the project influences the state legislative agenda remains to be seen.

Legitimacy: The design of the project recognized that experts, officials, stakeholders, and members of the general public all needed to be involved. The execution of the project saw environmentalists under-represented on the SC (by their own choice, because they distrusted the previous governor), although focus groups with environmental commissioners helped counterbalance the problem. Given an inadequate science base, the project did the right thing in refusing to deliver a single (but undefendable) ranking.

8. Conclusions

The impacts of environmental threats are unequivocally multi-dimensional. Some threats affect primarily human health, whereas others mostly affect the biota. Socioeconomic impacts often but not always derive from those two classes of primary impacts.

Public policy decisions frequently involve tradeoffs. Should we reduce human health impacts or ecological impacts or socioeconomic impacts with this marginal dollar of public funds? Yet the NJCRP demonstrates that the impacts of many environmental threats are uncertain and often highly variable across subpopulations. In

a strategic planning exercise where analytical resources are limited, it is extremely difficult to estimate impacts with precision, especially when there are multiple categories of impacts.

Thus the policy tradeoffs are uncertain. It is implausible that a formal tradeoff analysis will reveal a clear listing of policy targets ranked by net benefits or even by cost-effectiveness. The basis in scientific knowledge is simply inadequate.

Although formal analysis of some types of incertitude has become easier, as evidenced by the Monte Carlo analysis, remaining obstacles are severe. The Monte Carlo analysis does not address structural uncertainty or ignorance. Such problems often need to be addressed communicatively rather than instrumentally, when scoping the project, documenting and testing assumptions, and reporting the results. The procedural dimensions of the Sorting activity—establishing cut-points and identifying clusters—were more important than its analytical dimensions.

Given the incertitudes, it was appropriate for the NJCRP to seek to augment its legitimacy by procedural means. The Steering Committee sought civil legitimacy through its open deliberative process, its inclusion of lay public and stakeholder perspectives, its wide ranging multi-disciplinary review processes, its transparently written report, and its website with backup information.

Environmental decision makers inevitably proceed without complete information. They face deadlines and resource constraints that limit their ability to gather information, and in many cases, large gaps remain in the underlying scientific knowledge base. They must act when windows of opportunity open, or when the costs of waiting outweigh the benefits. These same decision makers, and the analysts who support them, invariably desire to have their actions seen as desirable, proper, and appropriate. After all, risk assessors and environmental decision makers often encounter hostile skeptics who question the appropriateness of methods and the propriety of decisions. Designers of future projects should therefore pursue legitimacy, through its civil and scientific sources, as the most likely route to effectiveness.

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A PROPOSED FRAMEWORK FOR MULTINATIONAL COMPARATIVE RISK ANALYSIS: PESTICIDE USE, IMPACTS AND MANAGEMENT

Report of the Comparative Risk Assessment Methods Workgroup

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Abstract

Comparative risk assessment is a natural tool for decision making regarding transboundary environmental issues. A workgroup of environmental experts met during the NATO Workshop over three days and addressed the state of the practice of comparative risk assessment and its applicability in an international context. Discussion of the application of comparative risk assessment led to a proposed case study on transboundary risks from pesticide use. Pesticides are of international concern because of transboundary trade of agricultural goods, impacts of applied pesticides on international environmental resources such as water bodies, differences in local and national safety practices, and concerns over the transferability of assessment models developed under different conditions than where pesticides are ultimately applied. Thus, a comparative assessment of potential impacts of pesticides on an international scale is proposed to be illustrative of the utility of the methodology for international environmental policy purposes. Workgroup discussions included the need to defining terminology, data needs, methods and tools for comparing risks, development of evaluation criteria, key issues regarding transboundary comparisons, and unique communication issues for international collaboration. A proposal for future work, including recommendations for international risk comparisons are also offered.

1. Introduction

Over the past fifteen years, comparative risk assessment (CRA) has emerged as a central tool in evaluating public health concerns, environmental management strategies and especially prioritizing environmental and ecological issues for communities and countries. In the United States alone, more than half of all states have conducted comparative risk exercises of sundry scopes and formats which frequently drive public policy and resource allocation within the public sector. Owing to its risk-based scientific ranking approach, CRA is widely used for identification of higher risk problems and setting priorities for research and action. CRA provides a powerful methodology to improve resource allocation, particularly if costs are explicitly introduced or risks are normalized for a given benefit. Policies based on comparative risk assessments could lead to more efficient use of resources and greater protection of public health and the environment. CRA can also be used for analysis and comparison of risks from two or more risk management alternatives that might be applied to the same environmental problem [1].

Increasingly, CRA is being used outside the U.S. but generally is applied within countries and even cities [2-4]. However, at the recent World Summit for Sustainable Development, comparative risk analysis was not only absent from the multi-lateral agenda that was manifested in the Johannesburg Declaration and Plan of Implementation, but also from the numerous bi-national and regional partnerships spawned by that international summit.

This stands in contrast with the increasing awareness that many environmental hazards require a multi-lateral intervention to be successfully addressed. According to one estimate, there are over 100 multi-national watersheds in the world. Because

wildlife are unaware of geopolitical borders, diversity protection often constitutes a regional challenge. Tropospheric ozone formation, acid rain and vehicle emissions are just a few of the issues in the area of air pollution which require a transboundary strategy, and could benefit from an international scale comparative assessment of risk.

Thirty-seven risk analysts and researchers from nineteen countries recently gathered in Anzio (Rome, Italy) under the auspices of NATO in a professional workshop to consider developments in comparative risk analysis in general and their applicability in international frameworks. This article offers the conclusions of a working group of experts that met at the workshop to this end.

The overall questions that we addressed are:

- How can comparative risk analysis be used in an international context in reaching a common set of environmental priorities and objectives?
- If so, what form, format, geographical scope and approach might be appropriate for such initiatives?
- What issues are important considerations?

To answer these questions we present a brief review of the range of methodologies presently utilized by risk practitioners and criteria for evaluating them within an international context. As data gaps pose a threshold obstacle to a multi-national comparative risk effort, information requirements are discussed with regard to their likely availability and the potential for models and other efforts to supplement empirical data when needed.

Finally, the group considered the example of agricultural pollution, in particular contamination and exposures from pesticides, as an example of an area in which regional or transboundary comparative risk assessment could aid in the identification of international priorities for sound pesticide management.

While other areas of interest, such as air pollution, might be more natural transboundary environmental media, efforts to consider a conventional, but concrete environmental problem such as pesticide applications reveal problems, both practical and conceptual, and demands which a multi-lateral comparative risk exercise may pose. At the same time, this evaluation suggests that multi-lateral risk exercises can lead to more efficient utilization of public resources by the participating nations, assist donor agencies in assembling a more cost-effective funding strategy, and most importantly, lead to greater environmental improvements as a result of logical allocation of resources and direction of energies in the environmental sphere.

In the remainder of the report we discuss prior applications of comparative risk assessment and available tools. We then present the results of our discussion on key international issues associated with the application of pesticides in a comparative risk framework. First, we define some key terminology.

2. Definitions

As in any international context, terminology may be used to express a range of ideas. Others have developed international glossaries of risk assessment terminology (e.g. Duffus [5]). Because we ourselves were an international group, we found it necessary

to agree on definitions of our key terminology. In this work, we define the term hazard as an event with adverse consequences, and risk as the probability of a hazard. Finally, we defined comparative risk assessment (CRA) as the simultaneous analysis, evaluation or ranking of multiple hazards and their associated risks.

3. Applications of Comparative Risk Assessment

Comparative Risk Assessment (CRA) provides a general framework for evaluating environmental problems affecting humans and ecosystems. Use of CRA for environmental problems started in U.S. with Environmental Protection Agency's (EPA) "*Unfinished Business: A Comparative Assessment of Environmental Problem*" report in 1987 [1]. The report, that evaluated more than 30 environmental problems in a comparative manner, reached the conclusion that the priorities of the environmental program at that time did not reflect the priorities determined by scientific methods.

Since then EPA has been promoting the use of CRA and related environmental planning tools by states, regions, cities, and native tribes to help communities in addressing their environmental concerns. Many comparative risk assessment projects and work for formulating methods to make broader use of CRA are currently in progress [7].

Outside the United States, U.S. Agency for International Development has commissioned about ten CRAs since 1990 for selected developing cities, countries, and regions in the world [2]. CRA has been also used in a number of developed countries. In Europe, both the EU and individual countries are working to adjust risk assessment techniques for application within their contexts [3]. CRA was employed in the preparation of the 1993 Environmental Action Programme for Central and Eastern Europe [4].

Reviews listing and comparing many CRA applications are available in the literature. The Green Mountain Institute for Environmental Democracy has a resource guide that lists hundreds of documents that reflect and/or discuss the various aspects of planning, implementing, and using the results from comparative risk assessment [8]. Morgenstern et al. [7] examines the experience with CRAs conducted in various developing countries in transition and compares both the methodologies and the results. The World Bank's Pollution Prevention and Abatement Handbook [4] also includes a summary of risk assessment projects in developing countries and transition economies.

A review of the first ten years of comparative risk assessment in U.S., primarily at the state and local level, is given by Jones [9] with the aim to provide an informed historical perspective on experiences with comparative risk approaches in legislative, regulatory, and policy contexts, and to address uses and misuses of these approaches. Comparative Risk Assessment Primer software, developed by Purdue University - Center for Technology Transfer and Pollution Prevention, includes summaries of 36 CRA projects conducted in U.S. and also detailed information on CRA methodology [10].

Several applications of comparative risk assessment, not listed in these reviews, are of interest. One example is the World Health Organization Global Burden of Disease study, which estimates disease and injury burden attributable to different

risk factors using the CRA methodology and models health outcomes from the distribution of exposure to the risk factors [11]. Another example is assessment of potential human exposures from the consumption of contaminated drinking water and related health risks. The results of such a CRA can provide useful information on the incidence and relative risks of different drinking water contaminants in a region; this analysis can be used to prioritize public health hazards and aid in the development of appropriate risk mitigation efforts [12]. Similarly, air pollution risks can also be evaluated using CRA [13].

Setting priorities for remediation of environmental contamination is another area where CRA can be extensively used. CRA can help to identify the best way to allocate financial resources for the cleaning up of the environmental contamination associated with industrial activities. It is also used to make informed decisions on the decontamination priorities of the sites, the extent of the remediation, and the techniques to be used for this purpose. The World Bank Environment, Industry, and Mining project in Bolivia, which includes CRA for the adverse effects of mining activities, like heavy metal contamination, acid generation, and physical hazards, is a satisfactory application in this area [14]. CRA can be used to evaluate the impacts of industrial waste disposal and the adequacy of and priorities for waste treatment policies can be highlighted, identifying attributes important to human, ecosystem health and to decision making for priority setting in the early stages of the environmental planning [15].

Agricultural pest management is a global, serious problem as approximately 50% of the world's food supply is destroyed each year by pests while the human population continues to expand rapidly [16]. Many potential advantages may be gained by including comparative risk assessment in the management of pesticides – including that CRA may result in increased commercial incentive to develop less hazardous products. In addition to this, data gaps may be filled in response to the commercial incentives to be able to demonstrate a product's lower risk in a comparative risk assessment [17]. CRA can be used also to compare the risks arising from chemical, organic, genetic engineering, and other pest management methods.

Comparative assessment of pesticides has already been shown to be effective and successful within several EU countries, such as Sweden. This principle is also included in Directive 98/8/EC concerning the placing of biocidal products on the market. Recently a report prepared by EC addressed the need to modify the directive concerning the marketing of plant protection products (pesticides) in certain respects and comparative assessment is listed as one area for consideration. Comparative risk assessment for pesticides features in the 5th European Community Environmental Action Programme (EAP) and is currently included in the proposal for the 6th EAP [18].

In addition to health risk assessments, comparative risk assessment can be used for ecological risk assessments. For example, CRA can be integrated in estuary management programs. The Lower Columbia River Estuary Partnership, a two-state public-private initiative, is a successful example of application of transboundary of CRA for priority setting. Using a watershed approach, the Estuary Partnership cuts across political boundaries, integrating 28 cities, 9 counties, and the states of Oregon and Washington in U.S. [19]. Another application area of CRA is marine environment risk assessment: the banning: the International Maritime Organization has developed a

global initiative that will eventually result in the ban of all antifouling systems exhibiting harmful effects on the marine environment. Given the number of alternative antifouling paints being developed, a process is necessary to determine the antifoulant expected to have the fewest impacts on the environment. CRA is a useful technique for such a purpose [20].

Rankings obtained from a CRA study may be useful for resource-allocation decisions. The risk-based process being introduced by the Department of Energy's Environmental Management Program at the nation's nuclear-waste sites is testing the effectiveness of translating the identification, analysis, and comparison of risks and remedies into budget decisions. The Commission encourages federal regulatory agencies to use comparative risk assessment for priority-setting on an experimental or demonstration basis [21]. CRA is also used in the energy sector. By applying CRA, the diverse characteristics, problems and requirements of energy technologies can be determined and the most efficient technology in terms of cost, public health and environment can be found [22]. There are also military applications of comparative risk assessment. CRA was used to assess different process arrangement alternatives and minimize the human health and ecosystem risks risk for managing and treating chemical agent stockpiles [23].

The remainder of this paper describes the discussions of the workgroup, and reports on key findings in on topics relative to the development of a proposed case study of the comparative risks of pesticide use and management in an international context.

4. Case Study

Pesticides are applied globally, but political and natural variation within nations affects their impact on health and the environment, both internally, and across ecological and trade barriers. The workgroup opted to address pesticides in an international context because of the international implications for food safety, for ecological impacts which do not follow political boundaries, opportunities for international cooperation on pesticide management, because of the range of expertise within the group in this area, and the opportunity to elucidate important factors in successful comparative risk assessment for the evaluation of international environmental issues. Further, the example made concrete the conceptual issues, such as the key data needs, comparative criteria, and key issues to be addressed.

Pesticides may pose transboundary problems when they are introduced into the environment. Because of their persistence in the environment, they can be transported through movements of waterbodies, air masses, or ocean currents. Some pesticides can be redistributed at a global level from warm-temperate to cold areas of the planet, and can be accumulated in aquatic or terrestrial organisms and transferred through the food chains. Pesticides may also pose a transboundary problem when contaminated food items such as fruits and vegetables are exported from areas of production to other areas and countries. Land use patterns and regional geography affect the migration of pesticides across political boundaries.

The remainder of this paper describes the discussions of the workgroup on topics relative to the development of a case study on the comparative risks of pesticide use and management in an international context. First, data needs and available and useful tools for comparing pesticide risks are described. Next, key criteria for comparing risks are evaluated. We then raise important issues for international risk comparisons, and suggest further work in this area. Finally, recommendations from the process are highlighted.

5. Data, Analysis and Tools

Numerous presentations during the four day workshop addressed important data issues and analytical tools useful for comparative risk analysis. For example, Schumann (this volume) discussed the effects of regional differences in soil type and irrigation/rainfall in Nepal, highlighting the need for local measurements of pesticides in addition to laboratory testing and models based on measures in other climates and soil types. Below types of data required for an international comparison of pesticides are discussed.

5.1. DATA NEEDS FOR COMPARATIVE RISK ASSESSMENT

For an international comparison of health and environmental risks from pesticides, data must be gathered from each source area. Many factors will affect exposure, and factors will vary by local practice and geography. Toxicity data is needed, but will not vary much across places. Specific types of data are needed to evaluate and compare exposure, such as residue levels in workers, food items, the environment, and more nationally specific data on tolerances, food consumption habits, importation/ exportation rates, worker standards, and historical information. Data must be of good quality and be comparable, measured with comparable apparatus and techniques. That is, the types of information must be on similar scale, frequency, aspect of pesticide use, and gathered by similar methods.

Data for evaluating pesticide risk across boundaries includes quantitative and qualitative information. Types of information include: biological, chemical, physical and environmental aspects of the system under consideration, socio-economic, political, and demographic data. The availability of data on pesticide use, residues, and exposure levels to workers and consumers varies across nations. National laws and customs dictate the gathering of information, and the level of safety with which pesticide use is managed. Gathering of new data for this study would improve comparability.

Data useful for comparing risks from pesticide use include identification of the sources of a pesticide, its characteristic features, environmental fate and possible adverse effects. Physical-chemical and fate and effect parameters may vary with environmental conditions. Data are needed on geographical/ climatic similarities and differences within and between countries, including: hydrogeology, meteorology, geotechnical properties, as well as land use, and local measurements.

Identification, geographical distribution, and classification of vulnerable ecosystems and exposed groups of populations is needed to compare risks across

political boundaries. This classification can help policy makers to make decisions on the areas and groups of population for which the measures have to be implemented.

Biomarkers could be used to compare exposures across pesticide exposed populations. Some biomarkers have been associated with specific environmental exposures. One workshop presentation, Attia, focused on the use of biomarkers to evaluate pesticide exposure in agricultural workers [24]. Biomarkers are used to indicate the presence of or level of activity of a given process, ideally, through the measurement of a reactant that is consumed or produced by that process, thus directly related to the mechanism-of-action [25]. Unfortunately, most biomarkers are not mechanistically derived but rather arise from an observed correlation with the existence of the process. At the genomic level, changes in the level of ribosomal nucleic acid or protein expression are markers that signify changes in biological activity, although they only comprise a single component of the overall process involved in determining biological activity.

Another workshop participant, Smirnova, discussed the use of mathematical models for identifying hypersensitive individuals chronically exposed to low levels of unfavorable factors (toxic substances, ionizing radiation and so on) [26]. To resolve this problem, the development of new approaches to risk assessment are needed, due to the ambiguity of effects of such exposures. In particular, hormetic effects were observed in a number of experiments with low doses of poisons and radiation. Therefore the new approaches must not ignore intrinsic properties of exposed organism. The implementation of such approaches calls for development and investigation of mathematical models describing mortality as an ultimate result of damage of mammalian organisms induced by an unfavorable factor. While the models developed were for evaluating risk from radiation exposure, biologically-based models could also be used to estimate risks for exposed individuals and populations in international comparative risk evaluations. .

5.2. TOOLS FOR COMPARING RISKS

Comparative risk assessment studies work best when conducted by multidisciplinary working groups including scientists, technical experts, industry representatives, government officials, citizens, and professional communicators, each having different backgrounds and viewpoints. Effective discussion and information flow both within and between groups are important for obtaining satisfactory results and meaningful comparative risk rankings. Advanced data visualization capabilities may facilitate this communication. Data in the form of a map is easier to interpret and communicate about than is tabulated data, and this affects the connections made and the conclusions drawn from it, particularly in a multilingual context. Drawing and layout tools are helpful in this respect and can be used to present the outputs from the risk assessment study to the decision and policy makers.

The first step of a CRA is the determination of the set of environmental problem areas to be analyzed and compared. For an international comparison of risks associated with pesticide use and management, a lot of information must be collected and analyzed. A non exhaustive listing of available tools that could be used in any risk

assessment study is presented, followed by few paragraphs specifically addressing tools for CRA.

5.2.1 GIS Models

The nature of any environmental or economic activity with a spatial dimension cannot be properly understood without reference to its spatial qualities. Therefore during the risk assessment process spatial dimension should be taken into the consideration. A Geographic Information System (GIS), with its advanced data integration, query, analysis, visualization and modeling capabilities, can be an effective and efficient platform for this purpose. Actually, GIS can be used as a tool for all phases of a CRA, including the determination of environmental problems to be addressed, scale of the evaluation, analysis of the risks they pose, and ranking them based on their relative importance.

A database management system, an integrated part of a GIS, provides means of rapid data access and query based on geographic location or attribute data. Using mapping functions of GIS, it is possible to superimpose two or more data layers and to relate otherwise disparate data on the basis of common geographic location. For example, exposed populations can be identified from a population density layer on a map identifying land uses. Also GIS makes it possible to explore and analyze data by location, revealing hidden patterns, relationships, and trends that are not readily apparent in spreadsheets or statistical packages. Because GIS products can be produced quickly, multiple scenarios can be evaluated efficiently and effectively.

5.2.2 Quantification Models

Numerous models are available for quantifying qualitative parameters (ecological factors, qualitative soil and pesticides parameters, etc.). Other workshop papers described a range of these tools, including Elbashyeb, Shatkin, and others, found elsewhere in this volume. These models include Fuzzy systems, Grey techniques, and other quantification techniques.

5.2.3 Statistical Models

Statistical models are used to improve and better understand subsequent patterns within the available data in order to be able to draw general conclusions. Statistical techniques can be applied for classification, estimation, prediction, clustering, and data description. It goes without saying that statistical models include all basic statistical techniques, such as probability distributions and inferencing methods.

5.2.4 Analysis Models

Interdisciplinary System Analysis models are useful when dealing with CRA applications due to the interdisciplinary nature of the system dealt with in such applications.

System simulation are valuable techniques when dealing with poorly understood systems, and where small changes in the system could have multiple big impact on the subsequent system and are usually irreversible. It is though useful to build simulation models in order to test different configurations, and choose the most

suitable one. Many CRAs rely on analyses with if/then rules, or expert judgment comparison.

In addition to these approaches, many of the review volumes discussed in the applications section also summarize available tools for CRA.

6. Comparative Decision Criteria

In a CRA, the criteria by which risks are compared may be based on risk to human health, ecology, or other endpoints such as impact on economy, or quality of life. For any comparison, the criteria must allow fair evaluation of each risk, and must be applied consistently to the data. The comparative decision criteria of an international CRA with a focus on pesticide use could evaluate affected populations at a regional, national or international scale. Broad participation by those affected by the comparison can help to ensure that important criteria are used for risk comparisons. Below, key criteria are discussed that should be considered in a multinational CRA. Section 7 highlights how the workgroup proposes to address these criteria.

It is not straightforward to determine how to compare different types of risks. For example, how does one weight risks to ecological receptors versus human health risk? Similarly, in characterizing the risk to human health, how does one compare different toxicological consequences such as cancer and non-cancer endpoints? Some of the decisions required to compare different risks are political rather than scientific, and therefore must include decision makers. The complexity of deriving comparative risk criteria increases if the comparison is across boundaries where many nations with differing priorities are involved. This section discusses options for comparing risks and the factors contributing to the complicated nature of risks comparisons.

6.1 EXPOSURE CRITERIA RELATED TO HUMAN AND ECOLOGICAL RISKS

It may be helpful to evaluate the relative distribution of any xenobiotic between the media or compartments (water, soil sediment, air and biota) at equilibrium [27]. Comparing the relative distribution of a contaminant can provide a good indication of the environmental compartment in which further CRA on effects should be conducted.

Given that the exposure of a foreign substance primarily depends on its fate in the environment, the way *mobility* (transport/translocation), *transformation* (metabolism) and *degradation* (mineralization/formation of irreversible bounds) are taken into consideration needs to be comparable as well because of the space and time scale implications of these three features that may change according to the type of source as well as the physic-chemical properties of a substance. This comparison should consider the relationship between dose or concentration of a substance and/or its metabolites and the incidence and severity of an effect on organisms or ecosystems.

6.2 COMPARING ECOLOGICAL RISKS

When assessing the ecological (ecotoxicological) hazard potential, an attempt must be made to weight the large number of different data required to yield an overall result that

will permit risks to be classified in an understandable univocal way. A useful way to weight, manage and sum up all these factors is the use of models to study and forecast the environmental fate of any xenobiotic.

A pragmatic approach (making use of the existing ecological knowledge) is needed to create foundations enabling the benefits and risks of a substance for human beings and the environment to be assessed [28] and overcoming the statistical uncertainty entailed by the many different ecological methods.

Comparing ecological models can be useful for CRA because of their iterative process of investigating real conditions, provided that assumptions and prerequisites on which models are based on are disclosed by the risk assessor. These are usually based on many different procedures and criteria that belong not only to the adopted modeling techniques but also to the risk assessment procedures of each country. Validation of the risk assessment procedures should be taken into consideration as well.

6.2.1. Effect of Bug Resistance

Tremblay [29] defines resistance as a drop of the sensitivity of any organism to a certain active substance. It is the outcome of changes occurred in the penetration, activation, degradation and excretion of usually a toxic active substance. Being hereditary, resistance to toxic substances is maintained by the next generations: usually the more frequently the same product is used at high concentrations on the same area, the sooner it sets up.

Bug resistance comparison criteria in the case of trans-boundary problems should pay attention not only to the kind of active substance together with the formulation but also to its rate of use and to the reproduction rate and mobility of the most dangerous bug.

6.3. TRANSPARENCY, SUSTAINABILITY, AND RISK PERCEPTION

Different types of risk (e.g. ecological versus human health) cannot be normalized to the same unit. Faced with this problem, the option is to clearly state the factors that affect the risk analysis so the comparison is made as transparent as possible. For example, institutional differences such as different countries having different methodological approaches, regulations and law (e.g. difference in analysis of pesticide problems, difference in national tolerance levels) cannot be expressed in any defined units but should be considered and clearly stated. The temporal and spatial scales should also be clearly expressed and if at all possible set to same scales for comparison. On a global scale of multidisciplinary CRA, where cross boundary comparisons of the risk in presence of political, geographical differences are inherent, clear explanation of these differences is most needed to normalize the comparisons to the extent possible. In cases of international comparison of risk assessments where the results cannot be easily normalized for direct comparison, the results can be analyzed within their context. In other words, for non-comparable units, the risks can be analyzed in parallel. For example, risks related to welfare, human health, and ecological health in different countries may be compared in their own categories as an alternative to translating the units in their own categories to a consistent unit, such as dollars.

Risks expressed in terms of cost may or may not be an advisable strategy. Using economics or another normalizing metric to compare the risks from different categories may create issues across boundaries, where cultural preferences create different weightings for categories of risk. If costs are used as a comparative criterion, these will have to be normalized with the participation of representatives from each study location.

Sustainability is a management criterion of risk assessment, and should be kept within this focus in order to be able to compare methods, tolerances, criteria, etc. In the case of pesticides, the sustainable management and assessment of risk are related to other criteria such as tolerance and perception. In order to achieve a sustainable risk management policy, a focus on the importance of the life cycle of pesticides, disposal management and the possibility of using auto-biodegradable products is suggested.

6.4. INCORPORATING VALUES DEMOCRATICALLY

Values are inseparable from comparative decision criteria. Yet, whose values should be used in risk comparison? Ideally, in comparing risks in a democratic society, the people who will be directly affected by the decision should make the comparison. A decision maker distant from the endpoints of the risk may have a limited basis for decision making. Effective public participation in decision making will pose a challenge in an international context and offers an opportunity for new collaborative work. Understanding people's preferences, and helping them form their own opinions which may temporally and regionally fluctuate can be difficult, even more challenging if different nations are involved. Many times, town meetings and public hearings may not be effective in educating people and eliciting people's opinions. For democratic decision criteria, significant progress is needed in risk communication.

Methods are needed for incorporating values in comparative decision criteria. Decision makers' and risk communicators' roles as facilitators for helping people form educated opinions and documenting their opinions should be more clearly defined to express this need. Considering that science develops in areas where there is a need or a demand, communicating this need is essential for potential development of a technique to scientifically incorporate decisions and values of people who will be affected by the risk. Perhaps, in the future, there may be a branch of science or trade that regularly records people's preferences and regionally and periodically updates a happiness scale similar to stock exchange indices. Such a happiness scale may incorporate Vermont's quality of life criteria such as aesthetics, economic well being, fairness, future generations, peace of mind, recreations, and sense of community [2].

7. Issues and Communication – A Case Study Proposal

In the development of a multinational CRA of pesticide use, impacts and management, a number of challenges must be addressed to allow for a coherent study that provides a clear understanding of current risks, and also serves as a guide for ongoing national and multinational data collection and risk management efforts. These challenges involve:

(a) the different temporal and spatial scales over which pesticide use and impacts occur;
(b) the high degree of variability in pesticide use and application practices and in the environments and populations that are impacted;
(c) differences in the way that data are collected and stored across different countries (and even *within* some countries), hindering their transfer and combination for a unified evaluation;
(d) identifying and obtaining input from decision makers that influence pesticide use and management in different nations, including regulatory authorities, intergovernmental agencies, agricultural aid and outreach organizations, manufacturers, distributors, and trade organizations; and
(e) ensuring the comparability of current and projected risk estimates across different health, safety and ecological endpoints.

The workgroup outlined a proposed approach for addressing these issues for evaluating pesticides in an international context. We do not expect that these issues can all be fully addressed in a single study. Rather, we propose to begin with a broad-scale effort that clearly delineates these challenges and measures the extent to which they are met, identifying the ongoing and future research, data-collection and management programs that could best assure improvements in subsequent assessments. Effective communication during the planning, implementation and dissemination of the study will be essential to ensure that progress is made in overcoming these challenges. The following summarizes some of the key features of these issues that our proposed study will address, as well as the communication methods that will be used to ensure its success.

7.1. SCALE ISSUES IN PESTICIDE ASSESSMENT

Pesticide use and impacts occur over a wide range of spatial and temporal scales. Risks of pollutant release to the environment occur during the manufacture and transport of compounds, during application, and following application to the field. Very near-field exposures and health risks occur first to workers involved in the manufacture and transport of pesticides, then (usually to a greater extent) to pesticide applicators in the field. These direct dermal and inhalation exposures can result in serious acute health effects to these workers. Subsequent dermal, inhalation and ingestion exposures can also occur following work hours to applicators and their families. These effects can be both acute and chronic for those who are involved in this activity over a period of many years. At a minimum, a survey of pesticide application practices and worker protection, education and exposure avoidance programs will be needed for countries included in a multinational study. This survey should be linked to available datasets on worker exposures, biomarker concentrations (in blood, urine, etc.), and documented health effects.

A second major route of pesticide exposure occurs to consumers due to their ingestion of chemical residuals in food. Depending on the type of food distribution network and the particular use pattern of the crops and foodstuffs (or other consumer products) made from them, exposures and impacts can range from local to national to multinational, over time scales of a growing season to a few years. The proposed study

will attempt to establish a source-receptor matrix for agricultural products for the nations included in the study, recognizing that some portion of the crops produced will be exported out of the region, and some portion of the food consumed will originate from exogenous imports. Within each country, the sub-categorization of locally-produced-and-distributed crops, vs. those distributed on the national market, will also be made. This will be done first for major grains, vegetables and fruits, with subsequent consideration of other products involved in the manufacture of more-processed foods, dairy and meat products (for an example of a methodology for estimating pesticide residuals in processed foods, see Hengel and Shibamoto [30]). The matrix will be linked to available information on trade and diet for each country, as well as a database of observed1 and allowable2 pesticide residuals in foods (this will allow a comparison of ingestion exposures and risks estimated under "current actual" vs. "current ideally-managed" conditions).

The final major route of pesticide exposure comes about through their general release and transport through the environment. This occurs primarily following application, but can also occur due to routine or accidental releases during manufacture, transport, or on-site formulation. The spatial scales of these effects can range from local or regional impacts on the ambient atmosphere, streams, groundwater aquifers, vegetation, fish and wildlife3, to the widespread global impacts now associated with persistent organic pollutants (POP's) and persistent bioaccumulating toxics (PBT's)4. Of the 12 pollutants now identified by international treaty as POP's requiring oversight and control, eight are pesticides (DDT, chlordane, endrin, heptachlor, mirex, toxaphene, dieldrin, and aldrin).5 The temporal scale of these impacts can range from months to decades, depending on the properties of the compound and the environmental receptors ultimately exposed. To provide a first assessment of risks resulting from environmental release, the suite of pesticides used in the study countries will be characterized in terms of their fate-and-transport properties for long-range transport, persistence and bioaccumulation. A number of recent multi-media environmental modeling assessments will be used to inform this evaluation.6 This analysis will be supplemented by the development of a database of reported pesticide concentrations in

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1 European Union reports on pesticide residuals in foods in selected countries are found at: http://europa.eu.int/comm/food/fs/inspections/fnaoi/reports/annual_eu/index_en.html. Examples of recent research studies on pesticide residuals in food include Lazoro et al. [31], Cabras and coworkers [32-35], Krol et al. [36], Saitta et al. [37] and Holden et al. [38].

2 Reports describing maximum residue levels prescribed for pesticides in food in the European Union are found at: http://europa.eu.int/comm/food/fs/ph_ps/pest/index_en.htm .

3 For examples of studies documenting these effects, see Qian and Anderson [39], Falandysz et al. [40], Meijer et al. [41], and Papastergiou and Papadopoulou-Mourkidou [42]. Observed datasets can also be complemented by the use of environmental fate-and-transport models, for example, Woodrow et al. [43, 44] and Barra et al. [45].

4 Documentation of global transport of pesticides is found, for example, in Cortes et al. [46] and de Wit et al. [47].

5 For more information on pesticides as POP's, see, for example:
http://www.sierraclub.org/toxics/factsheets/pops.asp
and
http://www.epa.gov/oppfead1/international/negotiation.html .

6 See, for example, Scheringer [48, 49], Bennett et al. [50,51], Eisenberg and McKone [52], and Hertwich and McKone [53].

water, soil, air, vegetation, fish and wildlife for the study region and adjacent areas. The effect of alternative application and field management practices on pesticide release to the environment will also be considered (e.g., Gan et al. [54]).

7.2 VARIABILITY IN PESTICIDE USE AND IMPACT

There is a high degree of nation-to-nation and site-to-site variability in pesticide use practices, regulatory policies and landscape conditions that affect the distribution of pesticides to food and different environmental compartments. To the extent possible, we will document these differences and include them in our assessment models in an explicit manner, using location-specific factors linked to a GIS map and database. We will also explore the use of statistical models to generate distributions of pesticide use; time-activity patterns for pesticide applicators and their families; meteorological, soil, and geo-hydrologic properties affecting environmental transport; resulting pesticide residuals and ambient concentrations; exposure factors for human and ecological receptors; receptor sensitivity; and subsequent health and ecological effects. These distributions will be used to supplement missing data at specific grid cells in the model, and also to represent system component variability within grid cells.

7.3 DATA COMPATIBILITY

Data on pesticide use, food residuals, ambient concentrations, exposures, biomarker concentrations, and health effects have been collected by government agencies and researchers in many countries. However, many of these data records are difficult to integrate because of differences in chemical names (and of course language), measurement methods, units in which the measurements are reported, the amount of information provided on the sample type and locations, and the degree of temporal and spatial averaging used in the measurements. We will attempt, when possible, to convert all data to a comparable basis and a common set of units. The data will be maintained in a common GIS database agreed upon by project participants during the first three months of the study period. We will determine whether protocols can be developed from those established in response to the July 1, 2002 European Union Communication 'Towards a Thematic Strategy on the Sustainable Use of Pesticides' (COM 2002-349, see: http://europa.eu.int/comm/environment/ppps/home.htm). We will also examine other cross-national studies of environmental health for possible data-reporting and archiving protocols.7

A particular concern will be to ensure that data collected on pesticide use, environmental conditions, chemical transport, food residuals, and human and ecological exposure factors can be interfaced at common spatial and temporal scales. To accomplish this, a single spatial unit will be selected for use in the GIS system and all

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7 Such as APHEIS: A European Information System on Air Pollution and Health (see: http://europa.eu.int/comm/health/ph/programmes/pollution/ph_poll_fp00_en.html), which includes cooperation by the European Environmental Agency, the Joint Research Centre, EUROSTAT and the World Health Organisation / European Centre for Environment and Health.

available data layers will be averaged (if originally at smaller scales) or disaggregated (if originally at larger scales) to match the spatial unit selected.

7.4 OBTAINING STAKEHOLDER INPUT

Effective participation and input is needed from the regulatory bodies, intergovernmental agencies, agricultural aid and outreach organizations, businesses, and trade organizations8 from across the countries participating in the proposed study. Project leaders from each participating country will be asked to identify and provide contact information for those stakeholders in their nation and region, and they will also be asked to establish a mechanism by which project study plans, information requests, and progress reports can be shared with them. Project groups will also be asked to characterize the flow of information and authority among their decision-making bodies and stakeholders, so that the institutional settings for pesticide management can be properly characterized for each participating nation.

7.5 COMPARABILITY OF RISK ESTIMATES

The assessment of risks from pesticides to workers, consumers, and the general population will include estimates for a significant number of human health, ecological and economic impacts. Two approaches will be explored for summarizing and presenting these results. First an attempt will be made to combine the different risk estimates for human health, ecosystem impacts, and economic effects into three aggregate measures of effect. Second, techniques for multi-attribute risk comparison and ranking will be used to explore the tradeoffs among these three major areas of impact.

The argument for seeking a common unit of impact for risks is that, while decision makers must address problems of different scale, nature and origin, action on these problems must be prioritized, ranked and compared to each other. In order to make such comparison, common units are required. Unless the common unit is achieved the problems cannot be compared and the decisions will be made instead based on perceptions and subjective opinions.

Although risk assessment cannot always eliminate subjective aspects, it is currently the best available tool to compare different threats and their consequences. The common unit that is the basis of the comparison is "risk."

A number of approaches have been explored for combining different risks to human health. One such approach is through the development of estimates of the "quality of life-years lost," which can be used to evaluate both premature mortality and morbidity effects [55]. Similarly, environmental and economic impacts may be evaluated using a common set of economic measures, such as the value of the ecosystem services lost as a result of the environmental damage and impairment, or the

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8 For example, through CropLife International and the Global Crop Protection Foundation, see: http://www.gcpf.org/

"willingness-to-pay" to avoid the ecological effect.9 Discussions among participating researchers and national and international decision makers and stakeholders will be conducted to identify the most pertinent impacts to include and the most appropriate methods for evaluating them. The discussions will then lead to the formulation of a final multi-attribute list of impacts for which risk estimates will be developed. Some of these estimates will be quantitative, while others will, due to limitations in available data and assessment methods, remain more qualitative in nature (for these, impacts will be estimated as either low, medium or high).

Risk assessment is a tool that pushes the point of comparison from the concentrations in the media to levels of risk at the receptors in threat. (See figure 1. in [57]) This give three unique feature to the risk assessment.

1. The decision is made not based on concentration levels but on risk levels.
2. This allows decision makers to compare and rank environmental threats of different environmental media (e.g., the threat of an air pollution problem with a groundwater contamination.)
3. The comparability of the risks is even wider, by providing comparable information not just between different environmental media, but between different types of hazards (eg. earthquake risks with risks due to flooding, or risks caused by chemical spills.)

8. Summary and Recommendations

The workgroup considered the practice of CRA in an international context, and reviewed applications, tools, methods, data needs, and issues regarding the implementation of a study to evaluate risks associated with the use of pesticides. We generated a proposal for an international CRA, to address methodological challenges and answer an important environment policy question: how best to address risks from regional scale activities within a nation that potentially impact human and ecological resources across political boundaries, and at variable scale.

In summary, we recommend: Data be gathered from available sources, and supplemented with original efforts to ensure comparability. Data needs for an international survey of pesticides include geophysical measures, residue measures, political information on practices and tolerances, and exposure measures reflective of the range of potential receptors (e.g. workers, consumers, ecological receptors).

8.1. TOOLS

Spatial tools, such as GIS, are an obvious choice for comparing risks among geographical units including countries and ecosystems, as well as numerous statistical tools, modeling approaches, and deliberation as discussed during the presentations of workshop participants and summarized in other papers in this volume. We propose

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9 A collection of different approaches for economic evaluation of environmental impacts is found in an April 15, 2000 special issue of the journal *Environmental Science & Technology*. The paper by Mourato et al. [56] specifically addresses the evaluation of health and environmental impacts from pesticide use.

incorporating many of these tools in an international comparison of risks associated with the use of pesticides on food crops.

8.2. ISSUES

Key issues for comparing risks include scaling issues, cross boundary variability, the need for comparable data, obtaining common metrics for comparison, and ensuring representative participation in decision making. One issue flagged for further development in our proposed international CRA is the need for approaches to compare data across endpoints, both in quantitative and qualitative terms. Other issues include the temporal relevance of such an evaluation, because any data collected reflect current activities, but need to be characterized by their past and future effects on exposed receptors. Our investigation proposes to address these issues.

8.3. BENEFITS

The workgroup members undertook this effort to identify resources because of a shared belief that CRA is a tool of potential import in the international environmental policy context. As international cooperation on environmental issues develops, tools can aid in identifying priorities, and analyzing alternatives. The risk framework offers a transparent way to consistently evaluate environmental practices with their associated human and environmental impacts across political boundaries. Comparative evaluations require consistent reporting mechanisms be developed, and fosters cooperation and communication among national representatives in decision making.

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Part 2

Environmental Decision Making

THE VALUE OF INFORMATION FOR CONFLICT RESOLUTION

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Abstract

A formulation for the value of information for conflict resolution is shown to provide insights and guidance for identifying the attributes of scientific procedures and studies needed to support participatory risk assessment and decision making. Traditional approaches to the value of information are first reviewed, including the determination of potential reductions in the uncertainty variance of risk-model outputs resulting from a proposed study or data collection program, and the economic value of information in a decision-analytic context. Limitations of these metrics are identified – when scientific assessments are conducted by multiple experts who may be exposed to either consistent or inconsistent observations, and when decision value is required for multiple stakeholders who may differ in their prior beliefs, methods for interpreting scientific studies, and their economic valuations for the outcomes of alternative decisions. Methods for identifying the sources and implications of differences in these among experts and stakeholders are presented. The use of a precautionary ratio is proposed as a means for characterizing the source of differing degrees of precaution exhibited towards a proposed project by different rational stakeholders, highlighting the programmatic and scientific changes that could be considered by project proponents to attempt to build a consensus with other, more-precautionary parties. Initial methods for computing a monetary value of information for conflict resolution are also presented.

1. Introduction

Democratic, participatory risk assessment and risk-based decision support are most effectively viewed as an iterative, deliberative process by which the multiple parties to a decision work together to (NRC, 1996):

i) formulate and frame the problem, viable options for addressing it, and the possible outcomes associated with different decisions;
ii) identify the current knowledge needed to understand and address the problem, as well as the additional scientific studies and data collection that could improve this knowledge;
iii) implement the planned studies and interpret the results and implications for the decision;

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iv) make a decision that satisfies, to the extent possible (consistent with legal and statutory requirements), the collective wishes of the stakeholders participating in the decision process; and

v) design and implement programs to monitor implementation of the decision and implications for scientific knowledge, future deliberation and decisions that could reinforce or modify the selected plan.

While demanding and ambitious from both a scientific and social-political perspective, such a program of action holds the best prospect for identifying risk management strategies that are both scientifically sound and acceptable to a broad set of constituencies.

To help implement this type of approach, a new perspective on the "value-of-information" is proposed. Traditional approaches to the value-of-information (VOI) are first reviewed, including engineering approaches based on the expected reduction in the uncertainty of predicted outcomes that could result from consideration of the information, and decision-analytic approaches based on the associated increase in the expected value of a decision to a single decision maker. A broader view of VOI – the *value of information for conflict resolution* (VOICR) – is subsequently introduced. The key factors that affect the VOICR are identified and implications are drawn as to how scientific studies for risk-based decision support could be designed to better ensure that it is maximized.

2. Traditional Approaches to VOI: Variance Reduction

Scientists and engineers often focus on the uncertainty variance of predicted outcomes from their assessments, and how much this variance might be reduced by new or additional data. This approach has been used, for example, by groundwater scientists modeling aquifer water elevations, flow rates, travel times and resulting contaminant plumes, as well the projected effects of alternative remediation options on these (Massmann and Freeze, 1987a,b; Loaiciga, 1989; Reichard and Evans, 1989; Cleveland et al., 1990, 1991; McKinney and Loucks, 1992; James and Gorelick, 1994; Wagner, 1995, 1999; Small, 1997; Sohn et al., 2000). These estimated model outputs are all uncertain as a result of uncertain subsurface conditions, in particular, the spatial profile of soil permeability or hydraulic conductivity in the aquifer, and other physical or geochemical properties of the system. Additional data can be collected with new wells to characterize the soils and/or groundwater, the analysis of additional chemical constituents in the groundwater, or the collection of more frequent samples. In addition, a broader set of laboratory or field studies could be considered to reduce either site-specific or general scientific uncertainties that affect the basic formulation of groundwater models and the accuracy and precision of their predictions. Similar options for data collection and additional scientific study are available in virtually all environmental and engineering domains where uncertainties in key model assumptions, formulations, inputs and resulting predictions prevail (e.g., Morgan and Henrion, 1990; Patwardhan and Small, 1992; Smith and French, 1993; Brand and Small, 1995; Abbaspour et al., 1996; Chao and Hobbs, 1997; Casman et al., 1999).

The basic computational procedure for predicting the uncertainty reduction resulting from new information is derived from Bayes rule:

$$\Pr[Event\ A \mid Data\ B] = \frac{Likelihood[Data\ B \mid Event\ A] \times \Pr^{o}[Event\ A]}{\Pr[Data\ B]} \qquad (1)$$

where $\Pr[Event\ A \mid Data\ B]$ denotes the probability of an event A, given the observation of data B. The possible set of data B may be generally viewed as the results of a diagnostic test, the data collected as part of a monitoring program, or the findings of a set of scientific studies. The probability of the event A, given data B, is referred to as a *posterior probability*, and it is computed from the prior probability for event A, $\Pr^{o}[Event\ A]$, and the probability that the data B will be observed given that event A is true, denoted as the *likelihood* function for the data: $Likelihood[Data\ B \mid Event\ A]$. The denominator of Equation 1 is the total probability of the data B, regardless of whether event A is true or not.

Consider now that event A is the value of an uncertain variable, X, with our uncertainty about X represented by the probability density function $f_X(x)$, where x is a particular value of X.[1] The data in Equation 1 are described by a test result or observation of X, O_X, however, this measurement is imperfect, with distribution $f(O_X|x)$. If the measurement is accurate, the mean value of $f(O_X|x)$ is equal to x: $E[O_X|x] = x$. If the measurement is precise, the variance, $\sigma^2_{O_X}$, of $f(O_X|x)$ is small. If O_X is obtained as the mean of many independent measurements of X: $x = x_i$; i = 1, n, then $\sigma^2_{O_X}$ is the error variance of each measurement divided by n: $\sigma^2_{O_X} = \sigma^2_{x_i}/n$. These characterizations of the accuracy and precision of the data are summarized by the likelihood function, $L(O_X|x) = f(O_X|x)$.

The observation O_X results in an updated, posterior distribution for the uncertainty in X, determined from Bayes rule:

$$f_X(x|O_X) = \frac{L(O_X|x)\ f^{o}_X(x)}{\int\limits_{u = all\ possible\ X} L(O_X|u)\ f^{o}_X(u)\,du} \qquad (2)$$

[1] To keep the conceptual development simple, *variability* in X is not considered; rather, X is assumed to have a single true value, but this value is unknown. For the case where X also exhibits variability (this will usually be spatial, temporal or individual-to-individual variability, for example in the exposure concentration or dose received by the different members of an exposed population – see, for example, Bogen and Spear, 1987; Hoffman and Hammonds, 1994; Cullen and Frey, 1999), the Bayesian updates presented here may be considered to apply to the parameters of the variability distribution, its mean, etc. (see, for example, Wood and Rodriguez-Iturbe, 1975; Iman and Hora, 1989; Small, 1994; Frey and Burmaster, 1999; Cullen and Frey, 1999; Gurian et al., 2001; Lockwood et al., 2001).

The Bayesian approach allows consideration of exogenous information and expert judgment in the formulation of the prior distribution for the uncertain quantity: $f_X^o(x)$; and combines this with the information of the observed sample O_X, to obtain the posterior distribution $f_X(x|O_X)$. Uncertainty updates can be made sequentially as new studies are completed and the data incorporated, with the posterior distribution of a previous update serving as the prior distribution for the subsequent analysis.

Methods for analytical or numerical implementation of Equation 2 are provided in Lee (1989), Gelman et al. (1995), Gamerman (1997), Kottegoda and Rosso (1997), Leonard and Hsu (1999), Monahan (2001) and DeGroot and Schervish (2002). In most cases, as more information is collected (e.g., in the form of a larger sample size, n, for $\breve{x}$), the uncertainty variance of $f_X(x|O_X)$ is reduced. In general, the uncertainty variance of a variable is apriori expected to decrease as more data are collected, though some data outcomes can cause the uncertainty variance to increase. These outcomes may be characterized as "surprises" that cause scientists to recognize possible processes and futures that they had either not previously considered, or had viewed as very unlikely (Shlyakhter,1994; Morgan and Keith, 1995; Casman et al., 1999; Hammitt and Shlyakhter, 1999).

Brand and Small (1995) illustrate methods for implementing Equation 2 for an integrated environmental health risk assessment, of the type shown in Figure 1.

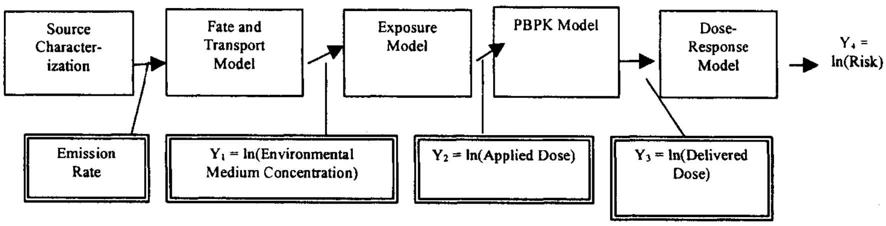

Fig 1. Integrated environmental health risk assessment model.

In this model, sequential calculations are made to determine pollutant emissions to the environment, resulting ambient and exposure medium concentrations, human or ecological exposure and uptake, the dose to targeted tissues or cells, and the resulting human health or ecological risk. In many cases (e.g., assuming linear fate and transport models, and linear, no-threshold dose-response models), the final risk may be approximated as the product of the set of single ratio terms for each submodel, including the ratio of the environmental medium concentration to the emission rate, the ratio of the applied dose to the environmental medium concentration, etc.. If the uncertainties in each of these ratio terms are lognormal (as is typical in environmental risk applications of this type), then the uncertainties in each of the intermediate model outputs, as well as in the final estimated risk, are also lognormally distributed. Brand and Small (1995) demonstrate implementation of this model using analytical or numerical Bayesian methods (the latter using Bayesian Monte Carlo, or BMC methods

– see, Dilks et al., 1992; Dakins et al., 1996; Bergin and Milford, 2000; Sohn et al., 2000) to compute prior estimates of uncertainty for each of the outputs of the model, as well as posterior estimates of uncertainty as new data are collected for each of the outputs of the integrated model. An illustrative result of this analysis is presented in Figure 2 (reproduced from Brand and Small, 1995).

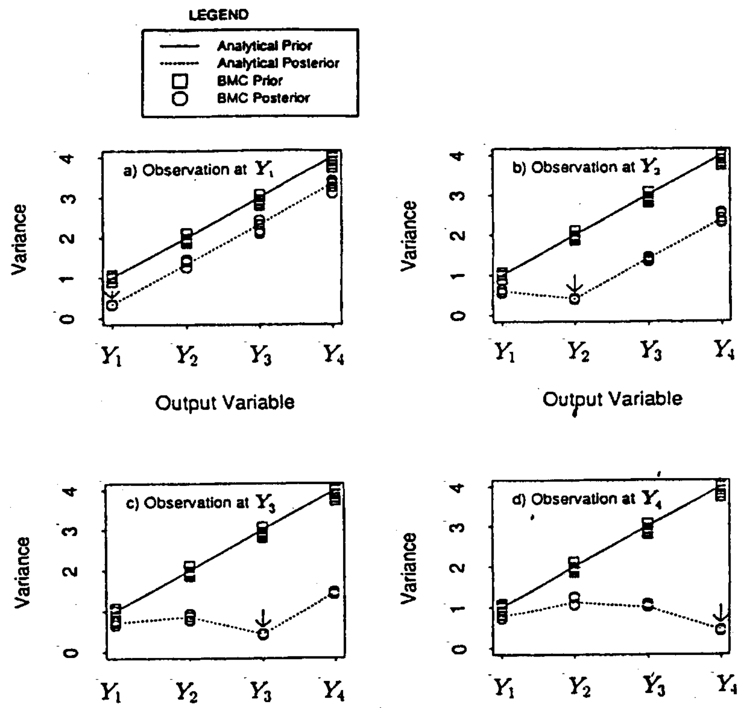

Fig. 2. Comparison of variance reductions obtained with the analytical and Bayesian Monte-Carlo methods for the first example. These reductions result from observations of the log outputs Y_k, for k = 1,2,3,4, each made with observation error variance $\sigma^2_{\epsilon k}$ = 0.5. The observed value was selected to be equal to the prior mean ($O_k = \mu_k = 0$ for all k). BMC results are shown for five replications, each with a sample size of 1000.

As shown in Figure 2, with equal variances assumed for the uncertainty of the ratio term representing each submodel, the prior uncertainty variances for the outputs of the sequential stages of the model increase linearly as one moves along the chain from emissions to ambient concentrations, exposures, doses and risk. Figures 2a-d show the uncertainty reductions that result from respective observations of ambient concentrations (Y_1), applied dose (Y_2), delivered dose (Y_3) and the risk or measured incidence in the population (Y_4), all assumed to be taken with the same likelihood function, or quality of information, as defined by the observation error variance, $\sigma^2_{\varepsilon k}$ = 0.5. The maximum variance reduction is achieved at the point in the model where the data are collected, with progressively smaller relative reductions propagating both forward and backward through the model. As such, if the objective is to minimize the uncertainty in the predicted risk, Y_4, the best place to collect information is at this point in the model (see Figure 2d).

However, the assumption of equal information value for the data collected at each stage in the process is rarely correct. Many environmental health risks assessments yield estimates of health effects in the affected population that are at rates well below those of the background incidence rate for the targeted endpoint. For example, background lifetime cancer risks in the human population are ~ order 10^{-2} to 10^{-1}, whereas many health risk assessments for toxic exposures attempt to identify and distinguish between lifetime cancer risks in the range of 10^{-7} to 10^{-4}. As such, the epidemiological "signal" is virtually impossible to distinguish from the background "noise". Animal toxicology studies usually have similar levels of uncertainty, due to the need to extrapolate from high to low dose, and from the test animals to humans. In these cases, more readily available ambient concentration data, exposure measurements, or biomarker data may yield the most valuable information and the largest uncertainty reductions in the predicted risk. A combined strategy of study and data collection at the various stages of the integrated environmental health risk assessment is thus expected to be most effective in most cases. Bayesian methods provide a means for identifying where these studies are likely to be of most value, and for combining the information and uncertainty reductions that result.

While the type of information shown in Figure 2 can be very valuable in depicting the current state of knowledge and uncertainty, and the potential for reducing this uncertainty with alternative studies, determining the uncertainty variance is in principle just the first step in characterizing information value. The key question is: In the context of pending risk management decisions, do the uncertainties matter? To address this question, the decision sciences have developed a methodological framework for VOI that considers: i) whether the reduced uncertainty could lead the decision maker to alter their decision; and ii) what the expected increase in the monetary value of the decision is as a result of the new information. This methodology is briefly reviewed in the following section.

2.1 THE DECISION-ANALYTIC APPROACH TO VOI

Decision analysis provides formal methods for choosing among alternatives under uncertainty, including options for collecting more information to reduce the uncertainty so that the outcomes associated with the alternatives are predicted with greater accuracy and precision (Raiffa, 1968; Keeney, 1982; Winkler and Murphy, 1985; Clemen, 1996; Chao and Hobbs, 1997). With no options for further study or data collection, the rational, fully-informed decision maker will choose the option that maximizes their expected value (or equivalently, minimizes their expected loss). Other decision rules may be considered as well, such as minimizing the maximum possible loss for a risk-averse decision maker, but all of these criteria should, with proper assignment of utility functions for outcomes, be convertible to the maximize-expected-value / minimize-expected-loss criteria.

When a possible program for further study or data collection is available, it should be chosen only if its results have the potential to influence the decision maker to change the preferred pre-information (prior) decision, and only if the increase in the expected value of the decision exceeds the program's cost. Since information of different types and different quality can be considered, and these can affect the uncertainty in the predicted outcomes associated with alternative decisions in different ways, a number of different measures of VOI can be considered (Hilton, 1981; Morgan and Henrion, 1990; Hammitt and Shlyakhter, 1999)[2]:

i) the Expected Value of Perfect Information (EVPI): how much higher is the expected value of the optimal decision when all uncertainty is removed?
ii) the Expected Value of Perfect Information about X (EVPIX): how much higher is the expected value of the optimal decision when all of the uncertainty about a particular aspect of the problem, X (e.g., a particular input to the environmental health risk assessment model), is removed?
iii) The Expected Value of Sample Information (EVSI): how much higher is the expected value of the optimal decision made contingent upon the results of a sampling or research program that has less than perfect information, that is, with finite sample size and/or the presence of some measurement error.

The EVPI is computed by determining the average value of the optimal decisions that would be made under each uncertain state of the world if it were known that this state were in fact true (the average value is computed by weighting each of the values associated with the different possible states of the world by the prior probabilities assigned to these states of the world) minus the expected value that is determined for the single optimal decision that maximizes the expected value considering the current level of uncertainty. The EVPIX involves a similar calculation, but assumes that only the uncertainty in X is removed when choosing the optimal

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[2] Even before these VOI measures are computed for alternative data collection programs, the decision analyst may wish to demonstrate the importance of considering uncertainty in the first place, by computing the Expected Value of Including Uncertainty (EVIU). The EVIU is the difference between the expected value of the deterministic optimal decision (derived from a deterministic analysis in which uncertainty is ignored) when considered under the more realistic conditions of uncertainty, vs. the expected value of the alternative optimal option that is selected when this same level of uncertainty *is* considered.

decisions for each of the (partially) informed states of the world. The EVSI requires simulation of the different research or monitoring program outcomes that could occur under each prior state of the world (again weighted by its prior probability) and the different optimal decisions that would be made once these different results are used to update the uncertainty. The EVSI thus requires a two-dimensional simulation of both the prior uncertainty for the states of the world and the different monitoring program outcomes that might occur with each.

3. Example Calculations of VOI

Examples demonstrating the computation of these different measures of VOI have been developed for environmental fate and transport models (Massmann and Freeze, 1987a,b; Freeze et al., 1990; James and Gorelick, 1994; Abbaspour, 1996; Wagner, 1999) and other elements of an integrated risk or economic assessment (Finkel and Evans, 1987; Taylor et al., 1993; Costello et al., 1998). In an illustrative analysis of an idealized sediment remediation program for New Bedford Harbor, Massachusetts, USA, Dakins et al. (1994, 1996) consider dredging programs designed to remove sediments contaminated with polychlorinated biphenyls (PCBs), so that PCB body burdens in fish will return to acceptable levels. The more sediment that is dredged, the more costly is the proposed remediation program. However, if an insufficient quantity of sediment is removed, the risk that PCB body burdens in the targeted fish will not return to acceptable levels is increased, thus increasing the risk of a prolonged fishing ban in the harbor and associated economic losses.

In the New Bedford Harbor example, Dakins et al. (1994) find that a dredging program chosen as optimal when uncertainty is ignored (to dredge 42,200 m^2 of harbor sediments), has a projected cost of ~\$42 million, but a significant risk of under-remediation, so that the expected loss due to a possible failure to recover the fishery resource in a timely manner is ~\$40 million and the total expected loss is thus 42 + 40 = \$82 million. However, the optimal remediation program when uncertainty is considered is to dredge 60,000 m^2 of harbor sediments, at a cost of ~\$60 million, but with an expected loss due to under-remediation of only \$2 million. The total expected loss is thus 60 + 2 = \$62 million, and the EVIU is 82 – 62 = \$20 million. Clearly, the initial decision to include uncertainty in the decision making framework has important economic (and risk avoidance) implications and benefits. Dakins et al. (1994) go on to compute the expected loss when there is perfect knowledge (i.e., complete uncertainty reduction) in the relationship between the amount of sediment removed and the resulting PCB body burden in the fish. As described above, the ideal amount of sediment is then assumed to be dredged for each of the prior dredge-area vs. PCB-body-burden relationships, with no risk of under- or over-remediation. The expected loss under this scenario (computed by weighting each prior relationship and associated sediment removal amount by its prior probability) is ~\$46 million. As such, the EVPI is estimated to be 62 – 46 = \$16 million. In a subsequent analysis, Dakins et al. (1996) calculate the EVSI for alternative site sampling and characterization studies that yield useful, though imperfect information on the site and the fish-sediment PCB

relationship, to range from \$5 million – \$15 million, depending on the size of the program and other model uncertainties not considered in the initial evaluation.

3.1.LIMITATIONS TO SINGLE DECISION MAKER VOI CALCULATIONS

The type of analysis described above, while very appealing in theory, has seen relatively little application in support of actual environmental management decisions. Part of the reason for this limited application is likely a result of the conceptual and computational complexities associated with this relatively new approach to VOI, but part of the problem may be even more fundamental. The basic decision model described above assumes a single decision maker with a single set of valuations for the outcomes, a single set of prior probabilities for these outcomes under the different decision options, and a fixed and known mechanism for translating study results into posterior probabilities, i.e., a known and agreed-upon likelihood function for the proposed or ongoing research and data collection.

However, in a democracy, few decisions of public import are made with such a uniform set of perspectives, values and beliefs. Rather, multiple stakeholders with different values and beliefs must deliberate and come to some consensus, informed by the science and the study results, but also affected by their differing valuations, prior probabilities and (as elaborated in the next section) likelihood functions. This often leads to conflict and stagnation in the decision process, or, when one party has the authority or power to impose its will on others, dissatisfaction by these other parties with the decision outcome. What is needed then is a decision analysis framework that identifies the sources of these differences and provides a rational basis for concrete steps that can overcome them. This leads to a broader and potentially more powerful notion of information value, based on the value of information for *conflict resolution*.

3.2. THE VALUE OF INFORMATION FOR CONFLICT RESOLUTION

The idea that better information could help to facilitate conflict resolution is an intuitive one. If part of the failure to reach consensus is due to a different view of the science – a disagreement over the "facts" – then a reduction in the uncertainty concerning these facts should help to eliminate this source of conflict.

Scientists often disagree on the facts (Cooke, 1991; Morgan and Keith, 1995; Hammitt and Shlyakhter, 1999). While the source of this disagreement may stem from ("legitimate") disciplinary or systematic differences in culture, perspective, knowledge and experience, or ("less legitimate", but just as real) motivational biases associated with research sponsorship and expectation, eventually, strong evidence that is collected, peer-reviewed, published, tested and replicated in the open scientific community and literature, should lead to a convergence of opinion. The Bayesian framework described above provides a good model for this process – even very different prior distributions should converge to the same posterior distribution when updated by a very large sample size with accurate and precise data. The perspective of variance reduction, in this case both *within* and *between* experts, may thus again be used to obtain an initial assessment of information worth for conflict resolution.

Stiber et al. (1999) illustrate this in the case of multiple experts interpreting data collected at a groundwater contamination site, attempting to determine whether reductive dechlorination of a chemical is occurring at the site. Reductive dechlorination is a first important step in the elimination from groundwater of a number of toxic chlorinated organic compounds, such as trichloroethene, TCE, by "natural means", allowing the use of the more-passive cleanup approach of "monitored natural attenuation," rather than a more-aggressive and expensive remediation option. Stiber et al. elicited 21 experts to develop a Bayesian Belief Network model for the occurrence of reductive dechlorination at a site for each expert, allowing computation of a posterior probability that reductive chlorination is occurring in the contaminant plume given the occurrence or non-occurrence of one or more of 14 pieces of evidence that can be ascertained from measurements at the site. These pieces of evidence involve both the *precursors* to reductive dechlorination at the site: the presence of reducing conditions, electron donors and proper environmental conditions for the temperature, pH and O_2 that promote its occurrence; as well as the appearance of the *results* of reductive dechlorination at a site: increases in the concentrations of chlorides and organic compounds that are reaction byproducts of reductive dechlorination.

Figure 3 shows the results of an evaluation of the effect of different types of evidence at a hypothetical site on the posterior probabilities that reductive dechlorination is occurring for the different experts. The first row shows the distribution of the prior probabilities for the 21 experts. As indicated, for the type of site described, the experts are initially quite divergent in their opinion as to whether reductive dechlorination will occur, with the most optimistic expert indicating a probability of 0.81 and the most pessimistic a probability of 0.14. In rows 2, 3 and 4 of Figure 3, the experts' priors are updated with the indicated positive evidence, supportive of the occurrence of reductive dechlorination. Depending on the conditional probabilities elicited for each expert, the different types of evidence affect each differently, though all indicate some increase in their calculated posterior probabilities with this positive evidence. With the four pieces of positive evidence shown in row 4, the mean probability that reductive dechlorination is occurring across the 21 experts increases from its prior value of 0.49 to a posterior value of 0.93, while the standard deviation across the 21 experts decreases from 0.20 to 0.13. A similar, but opposite effect is demonstrated for negative evidence in rows 5, 6 and 7. Thus, more evidence of a consistent nature (i.e., all positive or all negative) eventually leads the experts to converge in their posterior assessments.

However, as shown in the bottom three rows of Figure 3, conflicting evidence – some positive and some negative – causes the experts to *diverge* in their posterior probabilities, depending on the relative importance and weight that the different observations play in their belief networks. In all three cases shown, there is very little change in the mean probability across experts that reductive dechlorination is occurring, but a significant increase in the standard deviation (from the prior value of 0.20 to posterior values of 0.35 or 0.36). Indeed, the plots suggest that the conflicting evidence is able to split the experts into "two camps," those now more convinced that reductive dechlorination is occurring vs. those now relatively sure that it is not. While we hope that, at most sites, the evidence will be consistent, sometimes conflicting data will occur (especially if some scientists are simply incorrect in their understanding and

formulations for certain aspects of the problem). Still, it would not be surprising to find that for other problems, especially those that are in the early stages of scientific understanding and study, the initial evidence appears to be contradictory, leading to different inferences by different experts. Just as new studies and data are expected, on average, to decrease the uncertainty for a given expert, but could increase it, new studies are expected to increase consensus among multiple experts, but not in all cases.

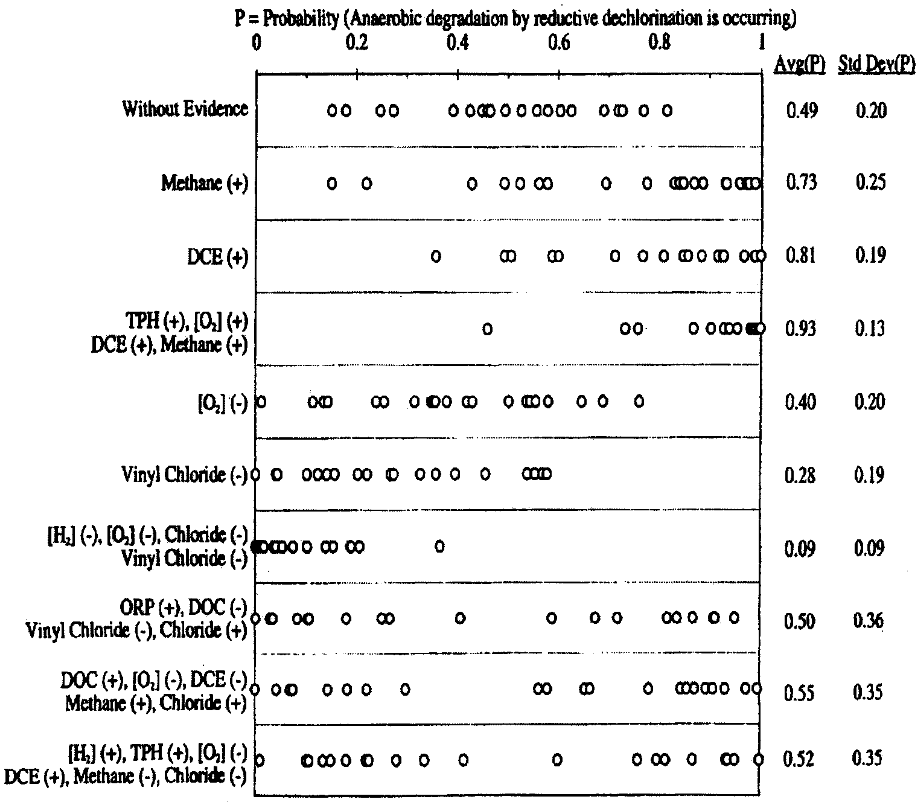

Figure 3. Distribution of expert model predictions for different cases of evidence. For each type of evidence, "(+)" is a positive finding and "(-)" is a negative finding.

4. Differences Among Stakeholders

Consider now a decision-analytic framework that must translate the implications of changes in assessments resulting from new information for scientists and the "decision-support community" into new assessments for decision makers and interested and affected parties. Even were the science to be perfect and all scientists and stakeholders agree that the outcomes associated with each decision option are known with certainty, the different stakeholders to the problem are likely to value these outcomes differently, due either to real differences in the allocation of the benefits, costs and risks associated with the program (Keller and Sarin, 1995), or due to different valuations assigned to the economic, environmental, social, political and ethical components of the decision alternatives and their outcomes, including interactions among these factors (Alhkami and Slovic, 1994; Cvetkovich et al., 2002).

Furthermore, the decision-making framework existing among the stakeholders – whether decisions are made jointly, unilaterally, and with or without collaboration – can affect both outcomes and information worth. In the classic prisoners' dilemma of game theory, shown in Table 1, two non-cooperating parties A and B will each choose option 2, even though it yields a lower return to them than would have occurred had each one chosen option 1, since the choice of option 2 increases their returns either way, once the other's choice is set. In contrast, if parties A and B cooperate and agree to decide together, then they could jointly agree to settle for the gains associated with option 1, yielding a $50 improvement for each compared to the joint selection of option 2.

TABLE 1. Prisoners Dilemma Problem from Two-Person Game Theory. The two non-cooperating parties will choose option 2, even though their payoffs are lower than they would have been had each chosen option 1.

Payoffs shown for (Party A, Party B)		Party B	
		Option 1	Option 2
Party	Option 1	($100, $100)	(0, $200)
A	Option 2	($200, 0)	($50, $50)

Similar problems arise in the "tragedy of the commons," wherein a common public resource in overused (or harvested too soon, i.e., "before the fruit is ripe") by non-cooperating agents. In the case of the prisoners' dilemma and the tragedy of the commons, the information to the parties is perfect, however, there are no institutional structures in place to implement the collective decision making needed to use the information to its best advantage. In the analysis that follows, a framework for considering the value-of-information for conflict resolution is proposed that is cognizant of these factors.

5. Conflict Resolution Among Stakeholders with Different Degrees of Precaution

As noted above, different stakeholders to a decision can exhibit different preferences among options due to different valuations of the outcomes and different probability distributions relating the options to the possible outcomes. The latter can be further dissected by noting that the different (posterior) probability distributions believed by the different parties to apply to the option-outcome relationship may arise due to i) different priors; ii) different information, that is, some parties may be privy to data or study results that others are not; and iii) different interpretations of the same information, that is, different likelihood functions. DeKay et al. (2002) use this framework to identify the attributes of a problem that could cause different rational stakeholders to exhibit different degrees of precaution when deciding whether or not to undertake an activity with uncertain benefits, risks and scientific studies suggesting whether the activity is safe or unsafe.3 While DeKay et al. view the problem in terms of differences in threshold probabilities for undertaking or eschewing the activity, the problem may also be couched in terms of a "precautionary ratio."

Assume Party A (call them the proponents of an activity) believes that the activity is safe with prior odds, $Odds_A{}^o(Safe)$:

$$Odds_A{}^o(Safe) = \frac{\Pr^o[A \textit{ believes activity is safe}]}{1-\Pr^o[A \textit{ believes activity is safe}]} \tag{3}$$

and perceives net benefits of $Benefits_A$ from the activity if it is safe and net costs of $Costs_A$ from the activity if it is unsafe. Assume further that Party A believes that the scientific studies and tests that are conducted to determine whether the activity will be safe have false positive (FP) and false negative (FN) rates of:

$$FPR_A = \Pr\left[A \textit{ believes scientific studies will indicate activity is unsafe} \mid \textit{Activity is safe}\right]$$
$$FNR_A = \Pr\left[A \textit{ believes scientific studies will indicate activity is safe} \mid \textit{Activity is unsafe}\right]$$

[3] The analysis of DeKay et al. considers items i) and iii) from the aforementioned list: differences among stakeholders in their priors and likelihoods, though not item ii) different information. Asymmetric information is often important as an impediment to equity and efficiency in economic trades, however, here we assume that all stakeholders receive the same data and information, however, they may interpret this information differently. As will be clear in the derivation and results that follow, the type of information that is under consideration usually involves studies "demonstrating a proposed program's safety," and the proponents of the plan have an incentive to ensure that this information is fully disseminated. The possibility that there may be a failure to share other information that could lead to the opposite conclusion (i.e., suggesting that the proposed activity is unsafe) is treated through assignment of a higher false negative rate for the information supporting the activity's safety.

Now assume that a second stakeholder, Party B (call them the potential opponents of an activity), has different prior odds that the activity is safe relative to Party A:

$$Odds_B{}^o(Safe) \;=\; \gamma_0\, Odds_A{}^o(Safe) \tag{4}$$

where the prior odds multiplier, γ_0, will in many cases be less than 1, since the potential opponents of the activity are likely to be more skeptical about the safety of these types of activities than are its proponents, even before scientific studies are conducted. Assume also that Party B perceives different costs from the activity, should it prove to be unsafe, and different benefits, even if proves to be safe:

$$\begin{aligned} Benefits_B(Safe) &= \alpha\, Benefits_A(Safe) \\ Costs_B(Unsafe) &= \beta\, Costs_A(Unsafe) \end{aligned} \tag{5}$$

Often it is expected that α will be less than 1, while β will be greater than 1, due, as noted above, to actual differences in the distribution of program benefits and exposures, health risks and other safety costs among the stakeholders, or differences in their valuations of the activity's benefits and risk impacts. Finally, the proponents and the potential opponents of the activity may have different perspectives on the likelihood function, i.e., the false positive and false negative rates of the studies conducted to determine whether the activity is safe, due to differences in their trust in the credibility of the studies.

It is widely recognized that scientists can often tend to overstate the confidence that should be placed in their inferences from scientific studies, due to overconfidence, omitted variables, etc. (Tversky and Kahneman, 1971; Kahneman, et al., 1982; NRC, 1996; Small and Fischbeck, 1999). A healthy skepticism by stakeholders is thus appropriate. Differential trust in scientific studies may also depend on who conducts them. If the studies are conducted by Party A, Party B may be suspicious, especially of the reported false negative rate, believing that Party A is more likely to report that the activity is safe when it is actually unsafe, due to conscious or unconscious oversights or dishonesty. Trust and resulting differences in likelihood functions can also be influenced by the extent to which stakeholders have input into the study plan, the level of outside, independent peer review, and previous experience ("they lied to us before, they are probably lying again"). As such, Party B's assessments of the false positive and false negative rates will often differ from those of Party A:

$$\begin{aligned} 1 - FPR_B &= \gamma_1 \left[1 - FPR_A\right] \\ FNR_B &= \gamma_2\, FNR_A \end{aligned} \tag{6}$$

where γ_2 will be greater than 1 whenever Party B believes that the studies were conducted either incompetently or dishonestly, relative to Party A; γ_1 will be less than 1 when Party B believes that the studies were incompetent[4]; and γ_1 will be greater than 1 when Party B believes that the studies were dishonest[5]. Empirical studies in recent years have helped to identify the factors that tend to build or diminish trust by the public in the competency and honesty of risk managers and their assessments of technological hazards (Slovic, 1993; Johnson and Slovic, 1995; Siegrist and Cvetkovich, 2000; Cvetkovich et al. 2002).

The net effect of these differences in beliefs and valuations by Party B relative to Party A are as follows:

If Party A requires posterior odds that the action will be safe of $Odds_A(safe)$ before they will support the activity (given A's values, priors, and interpretation of the evidence), then Party B will require that Party A in fact determines posterior odds of:

$$Odds_{A/B}(safe) = PR\ Odds_A(safe) \tag{7}$$

before they will support its implementation, where *PR*, denoted as the "precautionary ratio", is given by:

$$PR = \frac{\gamma_2 \beta}{\gamma_0 \gamma_1 \alpha} \tag{8}$$

The precautionary ratio provides a measure of the extent to which one stakeholder is more precautionary than another, and delineates the sources of this difference among the proponents and potential opponents to a project. As described above, the *PR* is likely to be greater than 1 (and perhaps much greater), since in many cases involving the proponents of a new technology, chemical product, hazardous waste incinerator, plan for nuclear waste disposal, etc., elements of the public who are more skeptical and precautionary than the proponents of the program are likely to:

- receive or perceive less benefits than the proponents, so that $\alpha < 1$;
- be exposed to or perceive a greater amount of risk should the plan prove faulty and unsafe, so that $\beta > 1$;
- have a higher prior probability that programs such as the one proposed are unsafe, so that $\gamma_0 < 1$; and
- entertain a higher false negative rate for the scientific studies purporting to show that the program is safe, so that $\gamma_2 > 1$.

Indeed, of the five factors included in Equation 8, only γ_1 is indeterminate as to whether a more precautionary public will believe that the scientific studies are more incompetent (so that $\gamma_1 < 1$, increasing the *PR*) or more dishonest (so that $\gamma_1 > 1$, thereby

[4] That is, the study is so incompetent that it cannot even determine that the activity is safe when it is safe.

[5] That is, since the studies "always conclude that the activity is safe," there is no need to be concerned about false positives.

decreasing the *PR*). The effects on the false positive rate are likely to be more subtle and smaller in any case, so it is expected that the effects of the other four contributing factors will dominate, and the *PR* will typically be >> 1 in most contentious applications where differing degrees of precaution are expressed.

The precautionary ratio depicts the key factors that typically inhibit the development of a consensus among stakeholders when considering contentious proposals with uncertain benefits and risks, shows the direct tradeoff between these factors, and helps provide guidance on the steps that could be taken to enable a cooperative, collective decision. If proponents of a program wish to convince a skeptical set of stakeholders of the worthiness of the proposal, they could:

1. ensure that the benefits of the program are more equitably shared, so that α is increased, becoming closer to 1;
2. ensure that the risks of the program, should something go wrong, are more equitably shared, with additional safety measures, insurance, etc., so that β is decreased, becoming closer to 1;
3. attempt to convince the public that their prior beliefs regarding the safety of the proposal are overly pessimistic, so that γ_0 is increased, becoming closer to 1; or
4. institute substantive and procedural changes to the scientific studies conducted to determine whether the proposal is safe, such as improving the measurements, increasing the sample size, including the stakeholders in the design and oversight of the study, ensuring that the studies are conducted or at least reviewed by independent third parties, etc., so that γ_2 is decreased, becoming closer to 1.

Approaches 1 and 2 are typically included as part of the negotiations between parties on the design and implementation of a plan (e.g., Susskind and Weinstein, 1980; Forester and Stitzel, 1989; Susskind et al., 2000), and proponents should always be open to innovative approaches and changes to a program that could enable these incentives and safeguards to be put in place (Gregory, et al., 1991; Kunreuther and Easterling, 1996; Smith and Kunreuther, 2001).

Approach 3, often viewed as the first and easiest thing for proponents to try ("if only the public was better informed and more rational . . . "), is now recognized to rarely succeed and is often counterproductive ("if they are trying so hard to convince us that this is safe, there must be something wrong, something they know that they are hiding . . ."). Instead the rational, precautionary public seeks assurances of safety of the type embodied in approach 4, where technical and institutional improvements are implemented to ensure the credibility of a negative finding (i.e., that the proposed plan is safe) by the scientific studies.

The insights from approach 4 have direct implications for the value of information for conflict resolution. The value of a proposed study plan or testing program emerges from both its actual accuracy and precision *and* its perceived credibility and trustworthiness by the key stakeholders. The design of scientific studies in support of risk assessments in a deliberative process must be cognizant of both criteria, and institute high-quality, credible programs to ensure that they are met. As noted in NRC (1996), these efforts are not independent, rather good science should

yield improved trust and deliberation, while effective participation, deliberation and input is expected to yield improved scientific insights and inferences as well.

5.1. AN ECONOMIC CALCULATION OF VOICR

The insights provided in the previous section can go a long way towards encouraging improved risk assessments in support of environmental decision making. While many of the conclusions are intuitive, presenting them together within a single framework of rational decision making should encourage a wider acceptance. Furthermore, as noted by DeKay et al. (2002), the very act of showing that precaution is not inconsistent with rational and effective scientific methods, and that risk assessment can be supportive of precautionary approaches, could provide the proponents of these different perspectives on environmental decision making with some common ground to explore (Jasanoff, 1993; Miller, et al., 1999). Still, the insights that can be culled from the development and exploration of a precautionary ratio are largely discrete, indicating what type of information is needed to allow for consensus development, but providing no direct measure of the economic value of this ability to enable consensus. In order to provide a link to the decision-analytic literature on economic measures of the VOI, a first exploration of the calculations needed to determine a monetary value-of-information for conflict resolution (VOICR) is provided. The approach builds on methods used in game theory (e.g., Fudenberg and Tirole, 1993), but adapts them in a manner that will be more familiar and accessible to those with experience in risk assessment. This first exploration is provided through an example problem illustrating the calculation of an expected value of perfect information for conflict resolution (EVPICR).

Consider three stakeholders, 1, 2 and 3, each of whom has assessed their uncertainty distributions for the net benefits (NB = benefits – costs) of a proposed program. Stakeholder 1 represents the proponents of the plan, Stakeholder 3 represents interests often, though not exclusively opposed to a plan of this type, while Stakeholder 2 represents the public or more-neutral stakeholders in the middle, supportive of plans of this type, but worried about the uncertain risks and associated costs. Assume that the assessed uncertainty distributions for NB are normal, with the parameter values shown in the second column of Table 2. A plot of these normal distributions is provided in Figure 4. Now assume that each stakeholder will support the proposed program iff: i) their expected net benefits are positive, that is, $E[NB] > 0$; and ii) their assessed probability that the net benefits are less than -100 is less than one percent, that is, $Pr[NB < -100] < 0.01$. This second criteria allows for a degree of risk aversion against a large loss that is common in technological and environmental decision making.

As shown in columns 3-5 of Table 2, Stakeholder 1 has both criteria satisfied and supports the proposed program. Stakeholder 3 rejects the proposal based on either of the criteria, both of which are unsatisfied. Stakeholder 2 rejects the proposal based on the second (risk aversion) criterion, even though the first criterion, requiring that the $E[NB] > 0$, is met.

TABLE 2. Computing the Expected Value of Perfect Information for Conflict Resolution

Column	Decision Criteria Under Prior Uncertainty				Computation of EVPI (when each stakeholder acts alone)			VOI for Conflict Resolution
1	*2*	*3*	*4*	*5*	*6*	*7*	*8*	*9*
Stakeholder	f°(NB)	E°[NB]	Pr°[NB]< -100	Support?	Pr°[NB>0]	E°[NB|NB>0]	EVPI	EVPICR
1	N(75, 50)	75	0.00023	Yes	0.9332	81.9	1.4	32.9
2	N(25, 100)	25	0.1057	No	0.5987	89.6	53.6	36.0
3	N(-25, 100)	-25	0.2266	No	0.4013	71.4	28.7	28.7
						Average:	27.9	32.5

The EVPI is computed for each stakeholder as the difference between the average value of fully-informed decisions (computed as the expectation across their respective prior distributions) and the expected value of their selection under the prior. With perfect information, each stakeholder opts to decline the program when NB < 0, and chooses it when NB > 0. With their current priors, the expected value of the fully-informed decision for stakeholder i is given by:

$$\begin{aligned} \textit{Expected Value Fully Informed}_i &= 0 \times \Pr_i{}^o[NB \le 0] + E_i{}^o[NB|NB>0] \times \Pr_i{}^o[NB>0] \\ &= E_i{}^o[NB|NB>0] \times \Pr_i{}^o[NB>0] \qquad (9) \end{aligned}$$

For stakeholder 1, the expected value of their selection under the prior is $E_1{}^o[NB] = 75$, while for stakeholders 2 and 3, $E_2{}^o[NB] = E_3{}^o[NB] = 0$, since they eschew the program under their priors. The EVPI for each stakeholder is thus given by:

$$\begin{aligned} EVPI_1 &= E_1{}^o[NB|NB>0] \times \Pr_1{}^o[NB>0] - 75 \\ EVPI_2 &= E_2{}^o[NB|NB>0] \times \Pr_2{}^o[NB>0] - 0 \qquad (10) \\ EVPI_3 &= E_3{}^o[NB|NB>0] \times \Pr_3{}^o[NB>0] - 0 \end{aligned}$$

Note that the second (risk aversion) criterion need not be considered when determining the effects of perfect information, since the net benefits become known to each participant with certainty (i.e., the distributions in Figure 4 collapse to a single point), so that the $\Pr[NB < -100] = 0$ whenever $E[NB] = NB > 0$.

For $NB \sim N(\mu, \sigma)$, the probabilities in Equation 10 are computed in the usual manner for normal random variables:

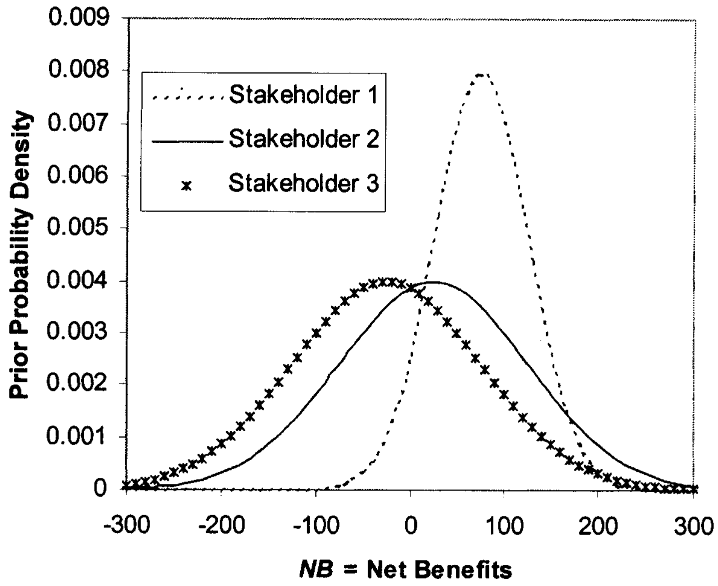

Fig 4. Prior Probability Distribution of Net Benefits for Stakeholder 1 (proponent), Stakeholder 2 (public or neutral stakeholder) and Stakeholder 3 (more inclined to be opposed to a project of this type).

$$\begin{aligned} \Pr[NB \le 0] &= \Phi\left(\frac{-\mu}{\sigma}\right) \\ \Pr[NB > 0] &= 1 - \Phi\left(\frac{-\mu}{\sigma}\right) \end{aligned} \qquad (11)$$

where $\Phi(\ )$ is the cdf for the standardized normal random variable, with mean zero and standard deviation 1. The conditional expected values are computed using relationships for a truncated normal distribution (Johnson and Kotz, 1970, Equation 79):

$$E[NB|NB > 0] = \mu + \left\{ \frac{f_{N(0,1)}\left(\frac{-\mu}{\sigma}\right)}{1 - \Phi\left(\frac{-\mu}{\sigma}\right)} \right\} \sigma \qquad (12)$$

where $f_{N(0,1)}(\ )$ is the pdf for the standardized normal random variable.

The results of Equations 11 and 12 are shown for each stakeholder in columns 6 and 7 of Table 2, yielding the estimates of $EVPI_i$ shown in column 8. Note that these

calculations assume that the each stakeholder is empowered to act alone, without consideration of the others. As indicated, the EVPI for stakeholder 1 is small (= 1.4), since this stakeholder is already so sure that the program is beneficial (resulting in their strong support for it under their prior), that the small probability that it is not is hardly worth exploring. In contrast, stakeholder 3 has a much wider prior variance than stakeholder 1, and since there is a significant chance (with probability 0.4013) that their prior choice to decline the program will change, their EVPI is larger (= 28.7). For stakeholder 2, the EVPI is larger still (= 53.6), since even under their prior they believe that the program is more likely than not to be beneficial (with probability 0.5987, and $E^o[NB] = 25$), but only decline to support it because of the risk-aversion criterion.

To compute the expected value of perfect information for conflict resolution (EVPICR), a decision-making rule must be assumed among the stakeholders. Here it is assumed that each of the three stakeholders must support the program for it to be implemented, so that each has veto power. As such, with the current uncertainty the proposal is rejected. The first modification that is made to Equation 10 in order to compute the $EVPICR_i$ is thus to assign the value zero to the last term in all three cases (rather than only for stakeholders 2 and 3 as currently shown in Equation 10).

The changes required for the terms $E_i^o[NB|NB > 0]$ and $\Pr_i^o[NB > 0]$ require further consideration. Since the value of a finding in support of implementing the program is computed for each stakeholder, it makes sense that the $E_i^o[NB|NB > 0]$ term should remain stakeholder-specific. However, since all three must agree that NB > 0 before a collective acceptance of the program can be implemented, we must calculate the $\Pr^o[NB_1 \cap NB_2 \cap NB_3 > 0]$. This calculation is complicated, since we do not know whether their perfectly-informed posterior assessments of the net benefits will remain different (i.e., they agree on what the outcome will be, but continue to place different valuations on this outcome), or will now, under perfect information, be the same (if they all place the same valuation on the known outcome). The pertinent question is, how much of the initial differences in the priors shown in Figure 4 are due to different expectations of the outcome, and how much is due to different valuations placed by the stakeholders on these outcomes? Clearly, a more careful construction of the prior distribution that disaggregates these factors, perhaps using the formulation presented in the previous section, would inform this issue (and this should be pursued in future development of the method). For now, we a make a simple, conservative assumption that the group is satisfied if and only if the most skeptical and precautionary stakeholder (in this case, stakeholder 3) is satisfied, and this occurs with apriori probability $\Pr_3^o[NB > 0]$. As such, Equation 10 is modified to compute the $EVPICR_i$ as follows:

$$\begin{aligned}
EVPICR_1 &= E_1^o[NB|NB > 0] \times \Pr_3^o[NB > 0] - 0 \\
EVPICR_2 &= E_2^o[NB|NB > 0] \times \Pr_3^o[NB > 0] - 0 \qquad (13) \\
EVPICR_3 &= E_3^o[NB|NB > 0] \times \Pr_3^o[NB > 0] - 0
\end{aligned}$$

The result of Equation 13 is shown in column 9 of Table 2. As indicated, compared to the *EVPI* when acting alone, the *EVPICR* for stakeholder 1 is much larger (increasing from 1.4 to 32.9), since the program is now declined under the prior, rather than accepted, resulting in a much greater potential for the prior decision to change compared to the case where stakeholder 1 acts alone. The EVPICR for stakeholder 3 is the same as their EVPI, while for stakeholder 2, their EVPICR is somewhat smaller than their (act-alone) EVPI, since they have a higher aprior belief than the decision-limiting stakeholder (3) that the net benefits will, with perfect information, turn out to be positive.

Consider now the collective value of the information across the three stakeholders. If this collective value is measured as a weighted sum of the value to each, then the average value represents the case where the assigned weights for the three stakeholders are equal. As noted at the bottom of Table 2, the increase in the EVPICR relative to the EVPI for stakeholder 1 is more than enough to offset the decrease for stakeholder 2, so that the average value of the computed EVPICR's across the three stakeholders is larger than the average value of their individual EVPI's.[6]

6. Discussion

This paper provides an overview of current approaches for assessing the value of information in a risk assessment, and begins to explore methods that could be used to address key barriers to conflict resolution and consensus building among diverse stakeholders. While a clear perspective arises on the need for scientific information to be generated in a competent, credible manner that is responsive to the needs of participants, the development of a general approach for implementing the methodology is still in its early stages. Further research will be needed to link the framework for characterizing information value in terms stakeholders' prior beliefs, likelihood functions and valuations for outcomes, as presented in the analysis of the precautionary ratio, with the preliminary monetary calculations presented in the previous section for the value of information for conflict resolution. As suggested by DeKay et al. (2002), empirical studies eliciting the inferences that stakeholders might draw from scientific studies conducted by different parties or with different designs would be particularly useful to learn more about the role of trust and credibility in affecting the way that such studies achieve value. Further exploration of theoretical methods from game theory could contribute as well.

An important aspect of future research will be the selection of actual case studies where the insights for collaborative risk assessment and decision making are clear and important. Many of the issues raised as part of this NATO Workshop on

[6] The very idea of computing an average value of the act-alone EVPI's is questionable, since the decisions made by each with perfect information often will not coincide, so that one stakeholder may accrue negative value while another benefits. This argues more strongly that the only appropriate way to compute the value of information across multiple stakeholders is in the context of the decision rules that apply to the group – in this case the three individual stakeholders cannot act alone, deciding to implement the single decision in question. The EVPICR thus provides an inherently more meaningful measure of multi-stakeholder information value.

Comparative Risk Assessment and Environmental Decision Making provide exciting possibilities for this type of exploration. The key environmental problems of the Mediterranean and Eastern Europe, such as regional air pollution from expanding industrial and transportation sectors and agricultural and food-source pollution associated with pesticide use, will require multinational efforts among a diverse set of scientists and stakeholders. It is hoped that the ongoing efforts to pursue these objectives by the participants in this Workshop can serve as a model of cooperation, both for these problems and others, and that some of the insights from this paper might help to further facilitate this process.

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INTEGRATED ASSESSMENT MODELING
A Simultaneous Equations Model of the Global Climate System

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Abstract

In this paper, we present a simultaneous equations model that provides a computationally efficient framework for computing long term, policy-dependent projections of global climate change. As part of our formulation, we explore the dynamic properties and numerical stability of the coupled system. We illustrate the framework with a numerical case study that utilizes the coupled system to compute projections of global-mean surface temperature change for three global carbon emissions control strategies. As part of our analysis, we explore the system's sensitivity to changes in the numerical specification of two key scientific uncertainties concerning the global climate system.

1. Introduction

Current efforts to confront the potential risks associated with climatic change present policy analysts, decision-makers, and intergovernmental negotiators with a host of challenges. In recent years, energy and environmental economists have focused attention on the development of a class of models commonly referred to as integrated assessment models of global climate change. Integrated assessment models (IAMs) are characterized by their broad-based approach to the analysis of the climate issue. Insofar as these models seek to represent the most salient features of the climate problem, they are typically comprised of analytically-tractable linkages between (i) models of atmospheric, oceanic, and biological processes; (ii) models of the global climate system; and (iii) models of the socio-economic processes that influence, and are affected by, climatic

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change.[1] Long-term projections of global climate change play a central role in most IAMs. General Circulation Models — by far the most sophisticated tools for performing global climate simulations — are ill-suited for the task of policy-oriented climate change assessment, in that the computational costs required to perform long-term simulations are largely prohibitive. In addition, large-scale climate models are unable to provide the degree of flexibility, ease-of-use, and transparency that policy-oriented modeling requires. For these reasons, policy-relevant assessments of global climate change necessarily entail trade-offs between model adequacy or realism, on the one hand, and computational efficiency, on the other. Recent integrated assessments of the climate issue have sought to make these trade-offs by utilizing *reduced-scale* models of the global climate system.[2] These computationally efficient models represent those processes that have the greatest influence on climatic change. While use of such models is common in IAMs that combine both economic analysis and climate science, little attention has focused on exploring the *behavioral* characteristics of these models when viewed as coupled, dynamical systems.[3]

In this paper, we present a simultaneous equations model of the global climate system, using a set of reduced-scale models that have been used in several recent climate-related IAMs. Our motivation for linking these reduced-scale representations together via a coupled system of equations is three-fold in nature. First, insofar as these models are often used to evaluate climate policy proposals, the manner and degree to which the system's variables interact with one another can have potentially important implications for how the overall system is estimated and interpreted. Second, exploration of the coupled system's formal properties provides a basis for better understanding the characteristic structure of the individual models that comprise the overall system. Finally, by exploring these issues in the context of an illustrative numerical example, we can better understand both the virtues and potential pitfalls of using frameworks like this to appraise control measures aimed at mitigating the potential adverse effects of global climate change.

The paper is organized along the following lines. Section 2 begins with a formal description of the coupled system of equations. We then explore the dynamic properties and numerical stability of the system. In Section 3, we utilize the coupled system to compute long-term projections of global climate

[1] For overviews of current approaches to climate-related integrated assessment modeling, see, e.g., Dowlatabadi [3], Parson [7], and Toth [9].

[2] See, e.g., Dowlatabadi [3], Nordhaus [6], Valverde [11], and Valverde, Jacoby, and Kaufman [12].

[3] The literature on this topic is somewhat scant. Braddock et al. [2] explore issues of equilibrium and stability in the context of the Dutch IMAGE model. More recently, Janssen [5] presents a non-linear dynamic model of the global climate system; the dynamical system is used within an optimizing framework that explores optimal emissions control strategies. Finally, Tucci [10] explores a related set of issues in the context of an econometric model of the world economy and climate system.

change for three illustrative global carbon emissions control strategies. We conclude, in Section 4, with a brief summary of our findings.

2. Coupled System of Equations

Table 1 summarizes a set of finite-difference equations used by Nordhaus [6] and others in several recent integrated assessments of the climate change problem. The four equations listed in this table represent simplified models for the global carbon cycle, CO_2-induced radiative forcing, and global-mean surface and deep-ocean temperature change. Specifically, C_t denotes the change in atmospheric CO_2 concentrations from its preindustrial equilibrium, E_t denotes anthropogenic CO_2 emissions, F_t denotes the change in radiative forcing corresponding to a volumetric concentration change from an initial concentration level at time period t_0 to a concentration level at some later time period t, and τ_t and τ_t^* denote the changes (at time t) in global-mean surface and deep-ocean temperatures, respectively. The parameters K_1 and K_2 denote the thermal inertias for land and ocean, respectively, and ν_d is the ventilation time of the deep ocean. Finally, τ_e is the e-folding or turnover time for the deep ocean, β is the marginal atmospheric retention rate, and λ is a feedback parameter.

At equilibrium, the feedback parameter, λ, is related to climate sensitivity and radiative forcing via the equation

$$\lambda = \Delta F_{2\times} \frac{\Delta}{T_{2\times}}, \tag{1}$$

where $\Delta F_{2\times}$ denotes the change in radiative forcing brought about by a static doubling of atmospheric CO_2 concentrations, and $\Delta T_{2\times}$ denotes climate sensitivity.[4] The equations shown in Table 1 are, for our purposes here, viewed as the structural equations of the global climate system, in the sense that each equation describes a particular facet of the climate system, and each is, in some measure, derived from first principles or physical theory.

In the absence of uncertainty, the four climate-related equations in Table 1 imply the following system of equations:

$$\begin{aligned}
\begin{pmatrix} C_t \\ \tau_t \\ \tau_t^* \end{pmatrix} &= \begin{pmatrix} \Gamma_{11} & 0 & 0 \\ 0 & \Gamma_{22} & \Gamma_{23} \\ 0 & \Gamma_{32} & \Gamma_{33} \end{pmatrix} \begin{pmatrix} C_{t-1} \\ \tau_{t-1} \\ \tau_{t-1}^* \end{pmatrix} \\
&\quad + \beta \begin{pmatrix} E_{t-1} \\ 0 \\ 0 \end{pmatrix} + \frac{1}{K_1} \begin{pmatrix} 0 \\ \rho(C_{t-1}) \\ 0 \end{pmatrix},
\end{aligned} \tag{2}$$

[4]Following Nordhaus [6], we assume that $\Delta F_{2\times}$ is equal to 4.1 Wm^{-2}.

$$
\begin{aligned}
C_t &= (1-\frac{1}{\tau_e})C_{t-1} + \beta\, E_{t-1} \\
F_t &= 6.3 \ln\left(\frac{C_t}{C_{t_0}}\right) \\
\tau_t &= \tau_{t-1} + \frac{1}{K_1}\left[F_{t-1} - \lambda\, \tau_{t-1} - \frac{K_2}{\nu_d}\left(\tau_{t-1} - \tau^*_{t-1}\right)\right] \\
\tau^*_t &= \tau^*_{t-1} + \frac{1}{\nu_d}\left(\tau_{t-1} - \tau^*_{t-1}\right)
\end{aligned}
$$

Table 1. Finite-difference equations for the global carbon cycle, CO_2-induced radiative forcing, and the globally-averaged two-box climate model.

where

$$
\begin{aligned}
\Gamma_{11} &= (1-\frac{1}{\tau_e}), \\
\Gamma_{22} &= -\frac{1}{K_1}\left(\lambda + \frac{K_2}{\nu_d}\right), \\
\Gamma_{23} &= \frac{K_2}{K_1 \nu_d}, \\
\Gamma_{32} &= \frac{1}{\nu_d}, \\
\Gamma_{33} &= -\frac{1}{\nu_d},
\end{aligned}
$$

and

$$\rho(C_{t-1}) \equiv F_{t-1} = 6.3 \ln\left(\frac{C_{t-1}}{C_{t_0}}\right).$$

In this system of equations, the variables C_t, τ_t, and τ^*_t are jointly dependent or endogenous, whereas the variable E_t is exogenously specified.

To simplify notation, we define a (3×1) column vector $\mathbf{y}_t$ and a (3×3) parameter matrix $\mathbf{\Gamma}$ as

$$\mathbf{y}_t \equiv \begin{pmatrix} C_t \\ \tau_t \\ \tau^*_t \end{pmatrix}$$

and

$$\mathbf{\Gamma} \equiv \begin{pmatrix} \Gamma_{11} & 0 & 0 \\ 0 & \Gamma_{22} & \Gamma_{23} \\ 0 & \Gamma_{32} & \Gamma_{33} \end{pmatrix}, \tag{3}$$

where the matrix elements Γ_{ij} in (3) are defined as before. In order to exploit the block form of (3), we partition $\mathbf{\Gamma}$ as follows:

$$\mathbf{\Gamma} \equiv \begin{pmatrix} \Gamma_{11} & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{\Gamma}_{22} \end{pmatrix},$$

where Γ_{11} is a scalar, $\mathbf{0}$ is a (2×1) zero vector, $\mathbf{0}^{\mathrm{T}}$ is a (1×2) transposed zero vector, and $\mathbf{\Gamma}_{22}$ is a (2×2) submatrix whose elements come from the lower right-hand corner of matrix (3), i.e.,

$$\mathbf{\Gamma}_{22} = \begin{pmatrix} \Gamma_{22} & \Gamma_{23} \\ \Gamma_{32} & \Gamma_{33} \end{pmatrix}. \tag{4}$$

Finally, combining the last two terms of system (1), we define a (3×1) vector $\mathbf{u}_t$ as

$$\mathbf{u}_t \equiv \begin{pmatrix} \beta E_t \\ \frac{1}{K_1}\rho(C_t) \\ 0 \end{pmatrix}.$$

Using these four definitions, system (1) can be expressed succinctly as

$$\mathbf{y}_t = \mathbf{\Gamma}\,\mathbf{y}_{t-1} + \mathbf{u}_{t-1}. \tag{5}$$

Equation (5) provides a succinct, structural representation of system (1). As we discuss below, this autoregressive representation provides a convenient means by which to explore the dynamic properties of the coupled system.

2.1 Dynamic Properties of the System

In exploring the dynamic properties of (5), we begin by noting that the system is valid for all values of t, in which case

$$\mathbf{y}_{t-1} = \mathbf{\Gamma}\,\mathbf{y}_{t-2} + \mathbf{u}_{t-2}. \tag{6}$$

If we now define the k^{th} power of the parameter matrix $\mathbf{\Gamma}$ as $(\mathbf{\Gamma})^k$ and, also, define $(\mathbf{\Gamma})^0 \equiv \mathbf{I}$, where $\mathbf{I}$ denotes the identity matrix, then substituting Eq. (6) into Eq. (5) and proceeding by induction, it is easily verified that

$$\mathbf{y}_t = (\mathbf{\Gamma})^t\,\mathbf{y}_0 + \sum_{j=1}^{t} (\mathbf{\Gamma})^{j-1}\,\mathbf{u}_{t-j}. \tag{7}$$

Equation (7) provides a computationally simple means by which to obtain numerical values of the vector time series $\mathbf{y}_t$. Here, $\mathbf{y}_t$ is construed as the sum of two components. The first component is $(\mathbf{\Gamma})^t \mathbf{y}_0$, which is a solution to the system $\mathbf{y}_t = \mathbf{\Gamma}\, \mathbf{y}_{t-1}$. In this way, the first component of Eq. (7) represents what $\mathbf{y}_t$ would be if it were influenced only by its own lagged values. As for the second component of Eq. (7), rearranging terms, we note that the difference

$$\mathbf{y}_t - (\mathbf{\Gamma})^t \mathbf{y}_0 = \sum_{j=1}^{t} (\mathbf{\Gamma})^{j-1} \mathbf{u}_{t-j}$$

can be interpreted as the combined effects of an exogenously-specified CO_2 emissions path $E_0, E_1, \ldots, E_t$ and the radiative forcing trajectory $\rho(C_0), \rho(C_1), \ldots, \rho(C_t)$ associated with this carbon emissions path.

In exploring the dynamic properties of Eq. (7), we begin by exploiting the fundamental structure of the parameter matrix $\mathbf{\Gamma}$. In particular, given its distinctive "block" form, it is possible to specify matrix decompositions of $\mathbf{\Gamma}$ which, in turn, allow useful inferences to be drawn about the dynamic behavior and stability of the overall system.

The decomposition of $\mathbf{\Gamma}$ has two parts. To begin, for submatrix $\mathbf{\Gamma}_{22}$, it is easily shown that if the eigenvalues of this matrix are distinct, then there exists a nonsingular (2×2) matrix $\mathbf{T}$ such that

$$\mathbf{\Gamma}_{22} = \mathbf{T}\mathbf{\Lambda}\mathbf{T}^{-1}, \tag{8}$$

where $\mathbf{\Lambda}$ is a (2×2) diagonal matrix with the eigenvalues, λ_1 and λ_2, of $\mathbf{\Gamma}_{22}$ along the principal diagonal and zeros elsewhere. Using decomposition (8), we can express the parameter matrix $\mathbf{\Gamma}$ as

$$\mathbf{\Gamma} = \begin{pmatrix} \Gamma_{11} & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{T}\mathbf{\Lambda}\mathbf{T}^{-1} \end{pmatrix}.$$

The diagonal structure of $\mathbf{\Gamma}$ necessarily implies that powers of this matrix are also diagonal matrices. In general, the k^{th} power of $\mathbf{\Gamma}$, $(\mathbf{\Gamma})^k$, is given by

$$(\mathbf{\Gamma})^k = \begin{pmatrix} \Gamma_{11}^k & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & (\mathbf{T}\mathbf{\Lambda}\mathbf{T}^{-1})^k \end{pmatrix}. \tag{9}$$

Given the nature of decomposition (8), powers of $(\mathbf{T}\mathbf{\Lambda}\mathbf{T}^{-1})$ are given by

$$(\mathbf{T}\mathbf{\Lambda}\mathbf{T}^{-1})^k = \mathbf{T}\mathbf{\Lambda}^k\mathbf{T}^{-1}. \tag{10}$$

Substituting Eq. (10) into Eq. (9) yields

$$(\mathbf{\Gamma})^k = \begin{pmatrix} \Gamma_{11}^k & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{T}\mathbf{\Lambda}^k\mathbf{T}^{-1} \end{pmatrix}.$$

Using the same diagonalization procedure described above, the parameter matrix $\mathbf{\Gamma}$ can be decomposed as

$$\mathbf{\Gamma} = \mathbf{SDS}^{-1}, \tag{11}$$

where the matrix $\mathbf{S}$ is a nonsingular (3×3) matrix and $\mathbf{D}$ is a (3×3) diagonal matrix consisting of the distinct eigenvalues of $\mathbf{\Gamma}$. For our purposes here, let

$$\mathbf{S} = \begin{pmatrix} 1 & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{T} \end{pmatrix} \qquad \mathbf{S}^{-1} = \begin{pmatrix} 1 & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{T}^{-1} \end{pmatrix}$$

and

$$\mathbf{D} = \begin{pmatrix} \Gamma_{11} & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{\Lambda} \end{pmatrix}.$$

Given decomposition (11), it follows that $(\mathbf{\Gamma})^t = \mathbf{SD}^t\mathbf{S}^{-1}$, in which case system (7) becomes

$$\begin{aligned} \mathbf{y}_t &= (\mathbf{\Gamma})^t \mathbf{y}_0 + \sum_{j=1}^{t} (\mathbf{\Gamma})^{j-1} \mathbf{u}_{t-j} \\ &= \mathbf{SD}^t\mathbf{S}^{-1}\mathbf{y}_0 + \sum_{j=1}^{t} \mathbf{SD}^{j-1}\mathbf{S}^{-1}\, \mathbf{u}_{t-j}. \end{aligned} \tag{12}$$

As in the case of Eq. (7), Eq. (12) provides a computationally efficient means by which to obtain values of $\mathbf{y}_t$ for specified values of t. It is important to note, however, that the formulation above assumes that the parameter matrix $\mathbf{\Gamma}$ consists of linearly independent eigenvectors. Of course, not all matrices are diagonalizable in the manner outlined above. To address this problem, we approach the decomposition of $\mathbf{\Gamma}$ from a somewhat different vantage point. As before, we focus attention on submatrix $\mathbf{\Gamma}_{22}$ of matrix $\mathbf{\Gamma}$. Using singular value decomposition, there exists orthogonal matrices $\mathbf{R}_1$ and $\mathbf{R}_2$ of order (2×2) such that

$$\mathbf{\Gamma}_{22} = \mathbf{R}_1\mathbf{\Lambda}\mathbf{R}_2^{\mathrm{T}}, \tag{13}$$

where $\mathbf{\Lambda}$ is a (2×2) diagonal matrix. The columns of $\mathbf{R}_1$ are eigenvectors of $\mathbf{\Gamma}_{22}\mathbf{\Gamma}_{22}^{\mathrm{T}}$; similarly, the columns of $\mathbf{R}_2$ are eigenvectors of $\mathbf{\Gamma}_{22}^{\mathrm{T}}\mathbf{\Gamma}_{22}$.

Using decomposition (13), the parameter matrix $\mathbf{\Gamma}$ can be expressed as

$$\mathbf{\Gamma} = \begin{pmatrix} \Gamma_{11} & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{R}_1\mathbf{\Lambda}\mathbf{R}_2^{\mathrm{T}} \end{pmatrix}.$$

As before, since $\mathbf{\Gamma}$ is a block diagonal matrix, the k^{th} power of $\mathbf{\Gamma}$, $(\mathbf{\Gamma})^k$ is given by

$$(\mathbf{\Gamma})^k = \begin{pmatrix} \Gamma_{11}^k & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & (\mathbf{R}_1 \mathbf{\Lambda} \mathbf{R}_2^{\mathrm{T}})^k \end{pmatrix}. \tag{14}$$

In contrast with the previous case, Eq. (14) does not factorize as simply as Eq. (9). To push the decomposition further, we impose an additional constraint on the matrices $\mathbf{R}_1$ and $\mathbf{R}_2$, namely, we require that

$$\mathbf{R}_2^{\mathrm{T}} \mathbf{R}_1 = \mathbf{I}.$$

Having made this assumption, it is easily verified that $(\mathbf{R}_1 \mathbf{\Lambda} \mathbf{R}_2^{\mathrm{T}})^k = \mathbf{R}_1 \mathbf{\Lambda}^k \mathbf{R}_2^{\mathrm{T}}$, in which case Eq. (14) becomes

$$(\mathbf{\Gamma})^k = \begin{pmatrix} \Gamma_{11}^k & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{R}_1 \mathbf{\Lambda}^k \mathbf{R}_2^{\mathrm{T}} \end{pmatrix}.$$

As before, the parameter matrix $\mathbf{\Gamma}$ can be written in spectral form as

$$\mathbf{\Gamma} = \mathbf{Q}_1 \mathbf{D} \mathbf{Q}_2^{\mathrm{T}},$$

where $\mathbf{Q}_1$ and $\mathbf{Q}_2$ are orthogonal matrices, and $\mathbf{D}$ is a diagonal matrix. For our purposes here, let

$$\mathbf{Q}_1 = \begin{pmatrix} 1 & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{R}_1 \end{pmatrix} \qquad \mathbf{Q}_2 = \begin{pmatrix} 1 & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{R}_2 \end{pmatrix}$$

and

$$\mathbf{D} = \begin{pmatrix} \Gamma_{11} & \mathbf{0}^{\mathrm{T}} \\ \mathbf{0} & \mathbf{\Lambda} \end{pmatrix}.$$

Since $(\mathbf{\Gamma})^t = \mathbf{Q}_1 \mathbf{D}^t \mathbf{Q}_2^{\mathrm{T}}$, Eq. (7) becomes

$$\begin{aligned} \mathbf{y}_t &= (\mathbf{\Gamma})^t \mathbf{y}_0 + \sum_{j=1}^{t} (\mathbf{\Gamma})^{j-1} \mathbf{u}_{t-j} \\ &= \mathbf{Q}_1 \mathbf{D}^t \mathbf{Q}_2^{\mathrm{T}} \mathbf{y}_0 + \sum_{j=1}^{t} \mathbf{Q}_1 \mathbf{D}^{j-1} \mathbf{Q}_2^{\mathrm{T}} \mathbf{u}_{t-j}. \end{aligned} \tag{15}$$

As in the previous case, for any admissible set of parameter values, Eq. (15) provides a computationally-efficient means by which to obtain numerical values for the vector $\mathbf{y}_t$.

2.2 Stability of the System

We now briefly explore the numerical stability of Eqs. (12) and (15). The stability properties of these systems is most easily seen by invoking a change of basis. Without loss of generality, we confine our attention to Eq. (15). Premultiplying both sides of Eq. (15) by $\mathbf{Q}_1^{\mathrm{T}}$ yields

$$\begin{aligned}\mathbf{Q}_1^{\mathrm{T}}\mathbf{y}_t &= \mathbf{Q}_1^{\mathrm{T}}\mathbf{Q}_1\mathbf{D}^t\mathbf{Q}_2^{\mathrm{T}}\mathbf{y}_0 + \sum_{j=1}^{t}\mathbf{Q}_1^{\mathrm{T}}\mathbf{Q}_1\mathbf{D}^{j-1}\mathbf{Q}_2^{\mathrm{T}}\,\mathbf{u}_{t-j} \\ &= \mathbf{D}^t\mathbf{Q}_2^{\mathrm{T}}\mathbf{y}_0 + \sum_{j=1}^{t}\mathbf{D}^{j-1}\mathbf{Q}_2^{\mathrm{T}}\,\mathbf{u}_{t-j}. \qquad (16)\end{aligned}$$

If we now denote the individual elements of the matrix $\mathbf{R}_2$ as

$$\mathbf{R}_2 = \begin{pmatrix} r_{11\,(2)} & r_{12\,(2)} \\ r_{21\,(2)} & r_{22\,(2)} \end{pmatrix}$$

and, as before, let $\mathbf{\Lambda} = \mathrm{Diag}(\lambda_1, \lambda_2)$, then $\mathbf{Q}_1^{\mathrm{T}}\mathbf{y}_t$ can be written explicitly as

$$\begin{aligned}\mathbf{Q}_1^{\mathrm{T}}\mathbf{y}_t &= \begin{pmatrix} \Gamma_{11}^t & 0 & 0 \\ 0 & r_{11\,(2)}\lambda_1^t & r_{21\,(2)}\lambda_1^t \\ 0 & r_{12\,(2)}\lambda_2^t & r_{22\,(2)}\lambda_2^t \end{pmatrix} \begin{pmatrix} C_0 \\ \tau_0 \\ \tau_0^* \end{pmatrix} \\ &+ \sum_{j=1}^{t} \begin{pmatrix} \beta\Gamma_{11}^{j-1} & 0 & 0 \\ 0 & \frac{r_{11\,(2)}\lambda_1^{j-1}}{K_1} & 0 \\ 0 & 0 & \frac{r_{12\,(2)}\lambda_2^{j-1}}{K_1} \end{pmatrix} \begin{pmatrix} E_{t-j} \\ \rho\,(C_{t-j}) \\ \rho\,(C_{t-j}) \end{pmatrix}. \qquad (17)\end{aligned}$$

Intuitively, Eq. (17) recasts system (7) in terms of the characteristic roots of the parameter matrix $\mathbf{\Gamma}$. By inspection, we note that a necessary stability condition for system (15) is that all of the eigenvalues of the parameter matrix $\mathbf{\Gamma}$ must be less than one in absolute value.

3. A Numerical Case Study

Numerical implementation of our coupled system requires that we specify the following data inputs:

1 Numerical estimates for the elements of the system parameter matrix $\mathbf{\Gamma}$;

2 An exogenous global carbon emissions time-path for each climate policy under consideration.

In what follows, we utilize the coupled system to compute long-term projections of global-mean surface temperature change for three illustrative global carbon emissions scenarios. As discussed below, we use alternative specifications for the parameter matrix, Γ, to explore the system's sensitivity to changes in the numerical specification of two key climate parameters. We look, first, at the climate change projections associated with a reference, or "business as usual," scenario, where no constraints are placed on global carbon emissions. For the purposes of comparison, we compute an analogous set of climate projections for a protocol proposed by the Alliance for Small Island States (AOSIS) and Germany [1]. In the version of the Protocol explored here, OECD countries agree to reduce CO_2 emissions to 20% below 1990 levels by the year 2005, and there are no commitments to reductions or limitations in greenhouse gas emissions by non-OECD countries.[5] Finally, we compute a set of temperature change projections for a *delayed* version of the AOSIS protocol, where the original target date is extended from 2005 to 2015.

For the first of the two data requirements outlined above, we utilize a set of parameter estimates reported by Valverde [11]. These estimates are derived from transient simulations of the MIT 2D-LO global climate model,[6] and they are indexed by two key climate-related parameters, namely, *climate sensitivity* ($\Delta T_{2\times}$) and *ocean diffusivity* (OD).[7] In Figure 1, we summarize the climate sensitivity and ocean diffusivity values explored here. The figure shows that climate sensitivity takes on three possible values: 1.5°C, 2.5°C, and 4.5°C, representing the lower, "best guess," and upper bounds, respectively, of the IPCC [4] climate sensitivity range; ocean diffusivity takes on the values 1 and 5, representing "slow" and "moderate" rates of warming.

This set of parameter values gives rise to six ($\Delta T_{2\times}$, OD) pairs, which, in turn, give rise to six separate specifications for the parameter matrix Γ. In specifying this matrix, we recognize that each transient simulation of the 2D-LO model is characterized by a fixed climate sensitivity value, from which we are able to derive — via Eq. (1) — a corresponding value for the feedback parameter, λ. The 2D-LO simulations of deep ocean temperature change assume that the deep ocean is 3,000 meters in depth. This assumption implies that $K_2 = 398\,\mathrm{J\,m^{-2}}\,K^{-1}\,\mathrm{yr}^{-1}$.[8] Most published estimates of the transient coeffi-

[5]The AOSIS proposal applies the 20% restriction to all the countries in Annex I to the Climate Convention, which include the OECD nations (except Mexico), plus 12 so-called "economies in transition" in the former Soviet Union and Eastern Europe. The percentage reductions do not account for leakage of emissions to non-OECD regions.

[6]For a technical discussion of the MIT 2D-LO global climate model, see, e.g., Prinn et al. [8].

[7]Climate sensitivity is defined as the difference in global-average surface temperature between equilibrium climates for current and doubled CO_2 levels; ocean diffusivity is a measure of the global climate system's rate of warming.

[8]A. P. Sokolov, MIT Joint Program on the Science and Policy of Global Change, Private Communication.

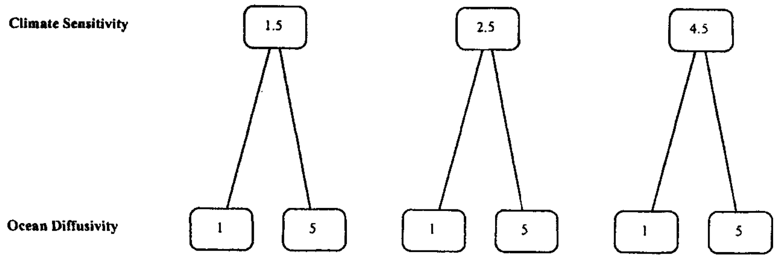

Figure 1. Climate sensitivity and ocean diffusivity values used in the coupled system projections of global-mean surface temperature change.

$\Delta T_{2\times}$	OD	$1/K_1$
1.5°C	1	0.015
	5	0.008
2.5°C	1	0.073
	5	0.009
4.5°C	1.	0.062
	5	0.015

Table 2. Estimates for the coupled system's inertial parameter, $1/K_1$, as a function of climate sensitivity and ocean diffusivity.

cient, ν_d, lie between 500 and 550.[9] Rather than assume a single value for ν_d, we implement two values, 590 and 118, depending on whether OD takes on the values 1 or 5. In completing our specification of Γ, all that remains is to specify a set of values for the inertial parameter, $1/K_1$, indexed by $\Delta T_{2\times}$ and OD. The estimates used here are shown in Table 2.[10]

Having specified numerical estimates for the coupled system's parameters and, also, having verified that these values satisfy the stability condition derived previously, we must now specify the global carbon emissions time-paths for the climate policy scenarios described above. For this task, we utilize the

[9]See, e.g., Nordhaus [6], who uses $\nu_d = 500$ in his DICE model.
[10]See Valverde [11] for a discussion of the experimental design and estimation procedures used to obtain these parameter values.

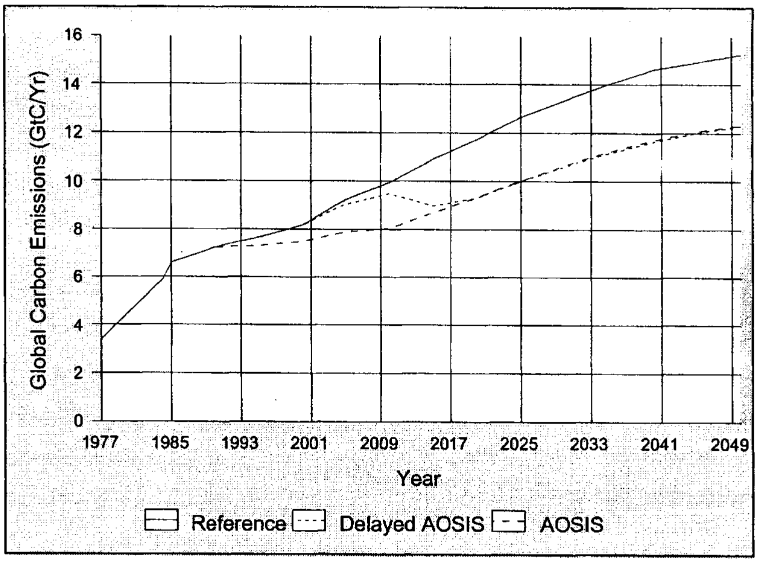

Figure 2. Global carbon emissions obtained from the MIT EPPA model for the Reference, AOSIS, and Delayed AOSIS policy scenarios.

MIT Emissions Prediction and Policy Analysis (EPPA) model,[11] which yields three global carbon emission paths (shown in Figure 2), one for each emissions control strategy.

Figures 3, 4, and 5 plot the coupled system projections of global-mean surface temperature change for the Reference, AOSIS, and Delayed AOSIS climate policies. Since each temperature change trajectory is characterized by a $(\Delta T_{2\times}, \mathrm{OD})$ pair, we are able to discern the influence that these parameters have on the resulting climate projections. What is, perhaps, most striking about the values shown in these figures is the countervailing influence that $\Delta T_{2\times}$ and OD have on projected temperature change. If, for example, we compare the projections for $(\Delta T_{2\times} = 1.5°\mathrm{C}, \mathrm{OD} = 1)$ with $(\Delta T_{2\times} = 4.5°\mathrm{C}, \mathrm{OD} = 5)$, we observe that the effects of high climate sensitivity are largely offset by the high ocean diffusivity value.

Our numerical implementation of system (1) also facilitates the making of pairwise comparisons of emissions control strategies. For example, Figure 6

[11]The MIT EPPA model is a global, computable, general equilibrium model that projects anthropogenic greenhouse gas emissions based on analysis of economic development and patterns of technical change. Documentation of the model is provided by Yang et al. [13].

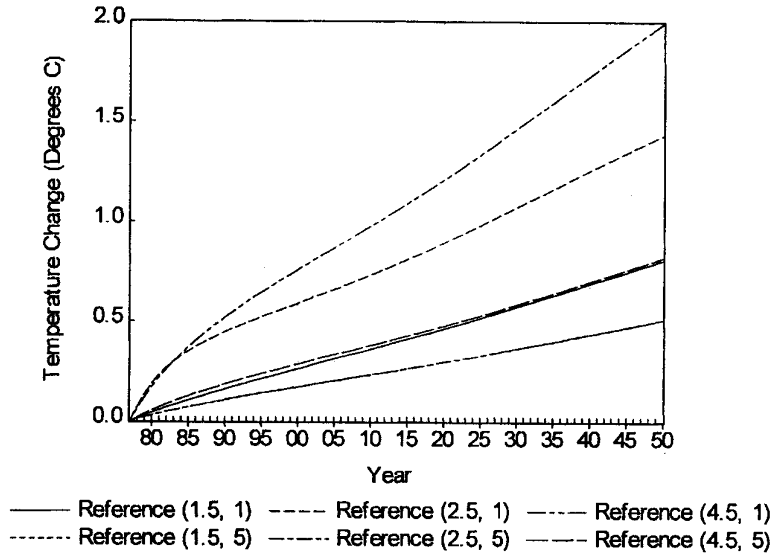

Figure 3. Coupled system projections of global-mean surface temperature change for the Reference policy scenario. Each temperature change trajectory is indexed by an ordered pair, ($\Delta T_{2\times}$, OD), which indicates the climate sensitivity and ocean diffusivity values used for that simulation.

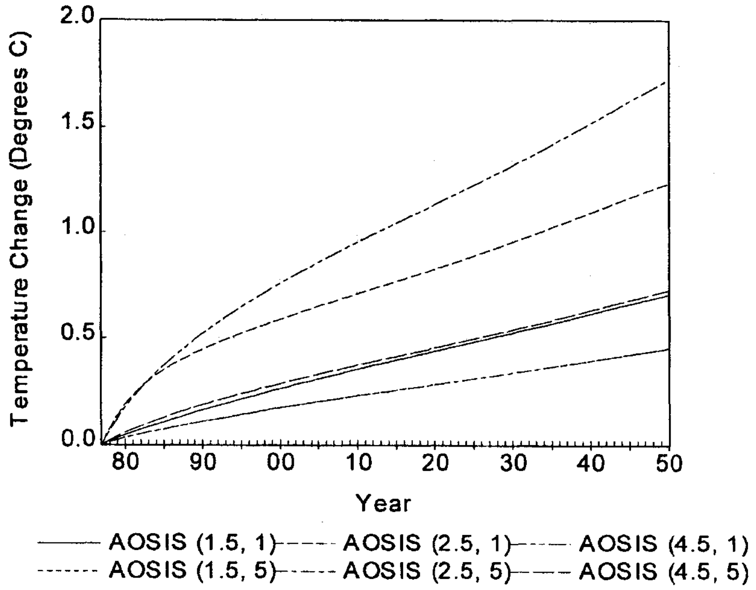

Figure 4. Coupled system projections of global-mean surface temperature change for the AOSIS policy scenario. Each temperature change trajectory is indexed by an ordered pair, ($\Delta T_{2\times}$, OD), which indicates the climate sensitivity and ocean diffusivity values used for that simulation.

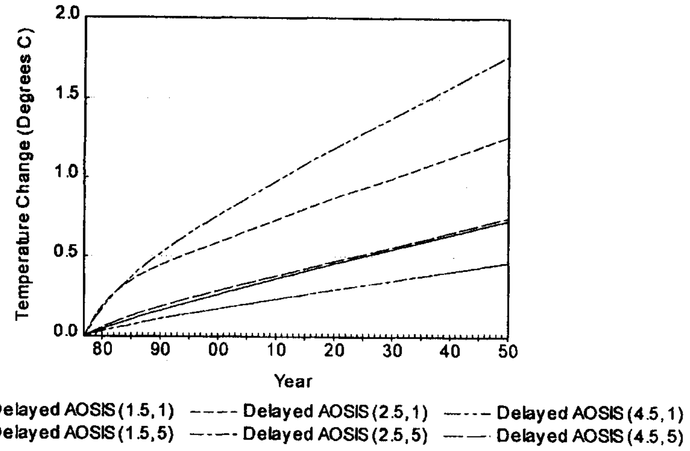

Figure 5. Coupled system projections of global-mean surface temperature change for the Delayed AOSIS policy scenario. Each temperature change trajectory is indexed by an ordered pair, ($\Delta T_{2\times}$, OD), which indicates the climate sensitivity and ocean diffusivity values used for that simulation.

shows the influence that OD has on the coupled system's projections of global-mean surface temperature change for the AOSIS and Delayed AOSIS policies. When $\Delta T_{2\times} = 4.5$°C and OD $= 1$, both climate policies give rise to rapid increases in temperature change; in contrast, when OD $= 5$, projected temperature change rises at a much slower rate, with a projected temperature change of 0.728°C and 0.747°C in 2050 for AOSIS and Delayed AOSIS, versus 1.726°C and 1.769°C for the OD $= 1$ case. We also note that for much of the 10 year period covering 2000–2010, there is considerable overlap in the projected temperature change trajectories for the AOSIS and Delayed AOSIS policies.

4. Conclusion

Long-term projections of global-mean surface temperature change are an important aspect of climate-related IAMs. The simultaneous equations model explored in this paper provides a nimble and computationally efficient means by which to obtain climate change projections for a broad range of carbon emissions control strategies. The mathematical representation used here allows for the characterization and evaluation of the behavioral characteristics and numerical stability of the system. In addition, our approach to numerically implementing the coupled system provides an analytical framework for exploring the manner and degree to which key scientific uncertainties such

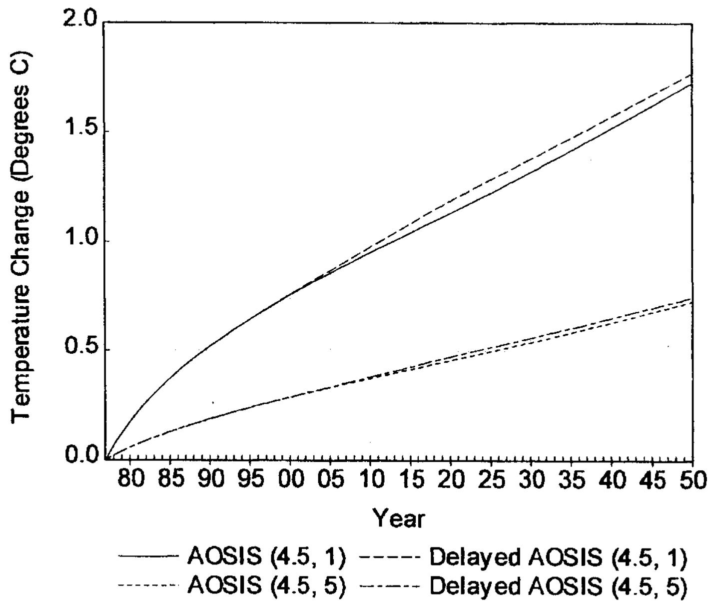

Figure 6. Comparison of the AOSIS and Delayed AOSIS climate policies, for $\Delta T_{2\times} = 4.5°C$. Each temperature change trajectory is indexed by an ordered pair, ($\Delta T_{2\times}$, OD), which indicates the climate sensitivity and ocean diffusivity values used for that simulation.

as climate sensitivity and ocean diffusivity influence long-term projections of climatic change. More generally, our approach seeks to provide energy and environmental economists with an instrumental means by which to enhance the realism of their integrated assessment modeling efforts, while still maintaining a reasonable balance between model transparency and computational tractability. Ultimately, such frameworks provide policymakers with a potentially useful means by which to appraise the uncertainties that underlie climate change projections, and, in addition, they provide a basis for evaluating competing climate policy proposals in light of these uncertainties.

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CLASSIFICATION SCHEMES FOR PRIORITY SETTING AND DECISION MAKING

A Selected Review of Expert Judgment, Rule-Based, And Prototype Methods

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Abstract

Agencies and organizations charged with priority setting require analytical approaches that are accurate, efficient, and reliable. Increasingly, decision analysis is applied using formal techniques that are measurable and repeatable. This paper surveys available methods ranging from expert judgment approaches to complex statistical models, and considers the benefits and issues raised for decision making that applies various approaches.

1. Decision Analysis

Decision analysis has emerged as an approach for structuring the process of setting priorities among alternatives in decision making. Some alternatives require risk/risk tradeoffs, while others may be associated with considerable uncertainty. Decision analysis offers a consistent approach to address alternatives and reduce the subjectivity of evaluating tradeoffs and uncertainty. Dozens of decision analytic approaches have been developed and tested, including some that simply structure the process of obtaining expert opinion, and others that include complex statistical models to weight and rank alternatives. Journals, listserves, and professional societies have formed in devotion to the topic. The purpose of this paper is to review and compare classification schemes, including Expert Judgment, Rule-Based, and Prototype Methods that have developed as formal methods used in environmental decision making.

Formal decision analysis is favored over informal methods because:

- Decisions are weighed based on transparent, predetermined criteria;
- Specific attributes important to decision-making can be compared (e.g., risk and cost); and
- Formal analysis can contribute objectivity to an otherwise subjective process.

However, formal analysis cannot convert a subjective process to an objective one. All approaches incorporate the values of the decision makers (e.g., in the selection and weighting of attributes) and must be reviewed by, and/or developed in cooperation with stakeholders and outside experts.

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Comparative Risk Assessment and Environmental Decision Making, 213–243.

Decision analysis has been applied in regulatory decision making in many U.S. agencies, including the U.S. Environmental Protection Agency (EPA) and in other governmental agencies. Some examples include:

- EPA's Office for Pollution Prevention, Pesticides, and Toxic Substances (OPPTS) Source Ranking Database, used to prioritize sources of indoor air pollution;
- EPA's OPPTS Waste Management Prioritization Tool, used to develop the Prioritized Chemical List; and
- EPA's Office of Solid Waste and Emergency Response's (OSWER) Superfund Hazard Ranking System.

Several of these approaches are discussed below.

2. Expert Judgment Methods

EPA routinely seeks expert advice when making important policy decisions. Expert judgment in priority setting usually occurs in the form of a workshop, or an advisory or expert committee meeting on a specific issue, for example, the 1997 Workshop convened for identifying microbiological contaminants for the first Contaminant Candidate List (CCL)[68]. The depth of knowledge gained from the experiences of these experts is far greater, more valuable, and more current than information that could be readily obtained by EPA through other sources such as relevant literature. However, the expert judgment rendered by the committees may be biased by professional affiliation, panel membership, composition and social pressures, as well as numerous other factors. The following methods are structured approaches which rely on expert judgment but attempt to eliminate some of the potential biases involved.

2.1 EXPERT ELICITATION

Structured elicitation of expert opinion is performed to achieve a rational consensus about an issue. The process is designed to produce reproducible, fair, neutral, and empirical results. The method begins with identification of the issue, identification of experts on the panel, and determination of the structure of the process. After elicitation of experts' opinions, a post-elicitation analysis is performed.

Variations of the expert elicitation model include the Paired and Classical Models. The Paired Model rates the relative importance of alternatives on various risk attributes. Alternatives are presented in pairs to the experts and each expert must decide which alternative of the pair is a greater priority. An alternative is compared pair-wise to every other alternative and then a final ranking is determined. The results can be evaluated for an individual expert or between many experts. The Classical Model of expert elicitation uses subjective probability to assess risk. Expert opinion is used to derive uncertainty distributions over model parameters. Opinions of each expert can be weighted equally, or weights can be derived from calibration questions which predict the statistical likelihood that an expert's opinion would be "right" or "wrong."

Expert elicitation was used by EPA to evaluate and improve a reductive dechlorination model for trichloroethylene [61]. The opinions of 22 experts were elicited using a structured interview process to establish probabilities associated with model variables. A graphical statistical approach called Bayesian belief network (BayesNet), reviewed in section 4.3.2 of this report, was used to improve the predictive ability of the model with the experts' information.

Expert elicitation was also applied to the assessment of surface water pollution impacts from a range of accident scenarios in Dutch industries [12] under the EC Seveso-Directive. A generic framework of accident scenarios was developed, and expert opinions of plant environmental managers and regulators were elicited to rank the importance of factors influencing the failure which results in an accident. The experts were asked to prioritize factors by pairwise comparison.

2.2 DELPHI PROCESS

The Delphi process is an iterative, consensus-building process that can be used as a generic strategy in making group-based decisions. An interest or expert group is typically assembled, either through correspondence (usually computer) or face-to-face discussion, to address options. Group members usually represent different points of view, and their opinions usually remain anonymous to avoid undue social pressures. After obtaining comments regarding a particular set of options, a facilitator analyzes the individual comments and produces a report documenting the group's response. Based on the discussion, panel members can anonymously alter their opinions in subsequent rounds. This process continues until the group reaches a consensus or stable disagreement. The final opinion of the group can be analyzed to provide a measure of variation within the "expert" opinion. Ordinal rankings can be provided as well as correlations between opinions. Multi-Dimensional Scaling (MDS) can provide information on the similarity of different variables (e.g., potential drinking water contaminants) on different axes.

Variations of the Delphi process include the Policy Delphi and the Trend Model. In the Policy Delphi, the group provides the strongest pros and cons for a number of differing selection criteria. This information itself can be valuable to decision makers or the group may use it to provide possible solutions to the policy makers. The Trend Model uses the panel to estimate a specific trend in an issue over a period of time (i.e., five years) including the uncertainties and assumptions that each expert used to estimate the trend. Subsequent iterations revisit the uncertainties and assumptions of the entire group, and the trend is then re-estimated.

EPA has used the Delphi process in decision making. For example, a modified Delphi process was used to forecast compliance with Information Collection Rule (ICR) monitoring by asking experts to identify the activities needed for compliance [70]. The Delphi process could be applied to set priorities among a set of potential drinking water contaminants, or to a list of contaminant attributes or properties.

3. Rule-Based Methods

Rule-based methods of classification weigh and combine attributes of the decision variables, in this case potential drinking water contaminants, based on a pre-determined algorithm. Rule-based methods include qualitative (e.g., categorical attributes are ranked), or quantitative approaches in which subjectively scored or physically measured attributes are developed and combined, generally in a linear way. Although they appear objective, the algorithms themselves and the data entered into the algorithms are generally a product of subjective expert opinion. However, unlike expert judgment, the observations can be objectively ranked and classified based on the subjective inputs. Below, a few types of rule based approaches are discussed, and specific approaches used primarily by the U.S. Environmental Protection Agency (EPA) are described. The methods range in objectivity from the qualitative approach under the California Safe Drinking Water and Toxic Enforcement Act of 1986 to the more objective and quantitative Cadmus Risk Index, discussed below. The rule-based methods also differ in their ability to deal with missing variables. Some, like the Cadmus Risk Index, do not include contaminants that have missing data. Myriad applications of rule-based prioritization methods exist in the open literature. Examples below methods developed with application to decision making for drinking water.

3.1 MULTI-CRITERIA DECISION ANALYSIS (MCDA)

MCDA is both an approach and a set of techniques, with the goal of providing a ranking of alternatives, from the most preferred to the least preferred option. The purpose of MCDA is to serve as an aid to thinking and decision making, but not as a rigorous tool that provides a final decision. As a set of techniques, MCDA provides different ways of disassembling a complex problem, measuring which options achieve primary objectives, and reassembling the pieces. MCDA combines quantitative and qualitative information into an impact analysis matrix (IAM) which provides a format to compare the rating of criteria versus the alternatives. Criteria are generally grouped into themes so that it is easier to get a general overview of the impacts [38].

Numerical decision analysis (NDA) uses numerical values of importance for each criterion. The general weighted average technique is most commonly used to compare alternatives. The most desirable criteria receive the highest score and the total desirability of any alternative is obtained by adding the criteria scores together. (The most desirable alternatives have the highest total scores.) The enhanced weighted average method uses individual fractional scores for groupings of related attributes (subcriteria) to obtain one score for a certain criterion. These fractional weights are largely subjective and unbiased expert opinions are needed to derive them. The weighted summation method uses a linear function to standardize quantitative scores and an overall score is calculated as the weighted average of the standardized scores. The Evamix method uses a mixture of quantitative and qualitative measures but this particular method produces results that are difficult to understand.

Verbal decision analysis (VDA) is a more qualitative method and uses verbal expressions of language to describe the quality grades of a criteria (low, medium, high). Pair-wise comparison of alternatives can also be achieved with VDA methods but do

not always lead to a clear alternative preference (e.g., dominance, equivalence, incomparability).

3.2 FUZZY LOGIC

Fuzzy logic is a way to structure imprecise information into models. Human decisions are made by incorporating all types of information, some quantitative, and other qualitative. Developing rules to process qualitative information requires a way to process data that is not immediately defined, such as the term tall, or toxic. Fuzzy logic allows qualitative attributes such as expert judgment to be addressed in rule-based and other quantitative classification methods. Fuzzy logic is an approach that seeks to extend Boolean logic to incorporate uncertainty. Rather than simple true/false outcomes, fuzzy logic applies "if, then" rules based on logical assignment rather than on precise values [10]. The idea was introduced by Dr. Lofti Zadeh of U.C. Berkeley to address the uncertainty associated with language in models. Fuzzy logic has been incorporated into neural network and other models for a range of applications including environmental control systems [11].

3.3. STRUCTURE ACTIVITY RELATIONSHIP (SAR) ANALYSIS

SAR and related structure based searches for chemicals are developed for prioritization of chemicals based on analysis of structural features. SAR was recently used in a joint effort by several offices of the US EPA to prioritize which disinfection by-products (DBPs) found in drinking water may be hazardous and need additional research [77]. This method was used to predict the toxic potential of chemicals with limited test information and data. This method involved using expert judgment to compare chemical structures of known carcinogens with the chemical structures of DBPs occurring in drinking water. The DBPs were then sorted into 5 categories from high to low levels of concern. Suitable candidates for further research were selected based on this ranking of concern level and general occurrence information. This method is quick and cost-effective way to prioritize a potential hazard.

3.4. EXAMPLES OF RULE-BASED APPROACHES

3.4.1. The Waste Minimization Prioritization Tool (WMPT)

EPA developed the WMPT in 1997 in the Office of Pollution Prevention and Toxics as a priority-setting tool for pollution prevention activities [67]. The WMPT is a rule-based decision analytical model, in which chemicals are ranked according to attributes contributing to the potential for human and ecological risk for persistent, bioaccumulative, and toxic chemicals (PBTs). EPA used the tool to prioritize source reduction and recycling efforts for 4,700 chemicals based on these attributes, or by definition, the Prioritized Chemical List. Attributes include human toxicity, ecological toxicity, bioaccumulation potential, persistence in the environment, and mass in the environment. For each chemical, attributes are scored from 1-3 (low, moderate, and high) representing a range of values for existing toxicity or physicochemical data. Scores are summed and the list ranked according to the potential for human health and

environmental risk. Exhibit 1 summarizes the information used in scoring chemicals for potential human health risk.

Human health risk potential score is derived from two sub-scores: toxicity and exposure. The human toxicity score is the higher of non-cancer and cancer toxicity scores. The human exposure score is the sum of a chemical's persistence, bioaccumulation potential, and mass scores. Because attribute scores generally represent scales of ten, when they are added to derive an overall chemical score, it is a multiplicative combination. The WMPT approach includes data of varying quality in the ranking. Similar to the scoring approach used by a National Research Council Committee [45], the most current peer reviewed information was first sought, and lesser quality sources used if no information was available from the first source. The short range of scores (1-3) may be a limitation for identifying differences in a chemical's potential for risk.

Shatkin and colleagues used the rankings from the WPMT as inputs to compare and prioritize treatment of hazardous industrial waste streams in Portugal [59].

3.4.2 Cadmus Risk Index (CRI)

The Cadmus Group, Inc. developed the Cadmus Risk Index in 1992 for EPA to identify and prioritize drinking water contaminants that may pose a threat to human health [63]. The pollutants are prioritized by the following hazard potential criteria: quantity produced, quantity released to water, frequency of detection in water, concentrations detected in water, persistence in water, and toxicity to human health.

The ranking scheme groups these criteria into three main factors that are combined to reflect the risk posed by a pollutant:

- *Production Factor (PF)* - factor is based on the quantity produced combined with human health risk (HR) of the contaminant and its persistence in water (PC);
- *Release Factor (RF)* - factor is based on the amount released in water combined with human health risk (HR) of the contaminant and its persistence in water (PC); and
- *Occurrence Factor (OF)* - factor is based on the occurrence in water combined with human health risk (HR).

Each factor and subfactor is scored based on available measured data. The scores are then adjusted to a scale ranging from 1 to 10 using the statistics program "Statview". The subfactors may be weighted based on their relative importance and combined to obtain the factor scores. The factor scores are then combined to compute the Risk Index for each pollutant using the following equation:

Risk Index (RI) = Production Factor score (PF) + Release Factor score (RF) + Occurrence Factor score (OF).

TABLE 1 Summary of Scoring Information for the Waste Minimization Prioritization Tool (WMPT)

Score	Human Toxicity	Ecological Toxicity	Bioaccumulation Potential	Persistence	Mass
low (1)	Non-Cancer Reference Dose >0.1 mg/kg-day or Cancer Slope Factor <0.01/mg/kg-day or low classification of alternative criteria	Sediment Quality Final Chronic Value (SQ FCV), Great Lakes Water Quality Initiative FCV (GLWQI FCV), Ambient Water Quality FCV (AWQC FCV > 10 mg/l, or low alternative criteria	Octanol/Water Partition Coeffecient (Log K_{ow}) <3.5, Bioaccumulation Factor (BAF) <250, or Bioconcentration Factor (BCF <250	Biodegradation time estimated to be less than 1 week	Actual or potentially releasable quantity, transformed to logarithm and divided by 2
medium (2)	Non-Cancer Reference Dose >0.001- 0.1 mg/kg-day or Cancer Slope Factor 0.01-1.0/mg/kg-day or medium classification of alternative criteria	SQ FCV, GLWQI FCV, or AWQC FCV = 0.1 - 10 mg/l, or medium alternative criteria	Log K_{ow} $3.5 and <5 BAF $250 and < 1000, or BCF=250 and < 1000	Biodegradation estimated to occur on a scale of weeks	Actual or potentially releasable quantity, transformed to logarithm and divided by 2
high (3)	Non-Cancer Reference Dose <0.001 mg/kg-day or Cancer Slope Factor >1/mg/kg-day or high classification of alternative criteria	SQ FCV, GLWQI FCV, or AWQC FCV < 0.1 mg/l or high alternative criteria	Log K_{ow}>5, BAF > 1000, or BCF > 1000	Biodegradation estimated to occur on a scale of months	Actual or potentially releasable quantity, transformed to logarithm and divided by 2

The chemical specific data used in this scheme was obtained from several U.S. governmental sources including: Integrated Risk Information System (IRIS), STORET, Pesticides in Ground Water Database (PGWDB), HAZDAT, Permit Compliance System (PCS), Toxic Release Inventory System (TRIS), and Hazardous Substances Data Bank (HSDB). This ranking scheme allows for the assessment of large amounts of data.

The Cadmus Risk Index uses both exposure and toxicity information to develop the ranking system and can use information from many different data sources. Toxicity scores include a weight of evidence classification. For example, the carcinogenicity score is developed by weighting the score for the Unit Risk value with a score from EPA's carcinogenicity classification. Non-carcinogens are weighted by a severity of health effect score. However, in the Cadmus Risk Index, if a critical data element is missing from a data source, the contaminant will not be included in the ranking scheme.

3.4.3 AWWA Screening Process

The AWWA Screening Process, developed by the American Water Works Association, uses chemical toxicity and occurrence information in addition to technical and economic information to determine the economic feasibility of the regulation of specific contaminants in drinking water. This scheme is not applied to a general list of chemicals, but is applied to a particular contaminant (or group of contaminants with similar characteristics) found in water.

In this method, data quality is considered and contaminants with data gaps can be evaluated. Exposure data was obtained from the following sources: National Organics Monitoring System, National Pesticide Survey, STORET, the Federal Reporting Data System and U.S. Geological Survey databases. Toxicity information was obtained from government databases (IRIS, Health Effects Assessment Summary Tables, Agency for Toxic Substances and Disease Registry, HSDB, Registry of Toxic Effects of Chemical Substances, Chemical Carcinogenesis Research Information System, and Developmental and Reproductive Toxicology Database) and provides a qualitative measure of the chemical's existence, quality, and applicability. The qualitative nature of the process allows for compounds with missing information to be included.

For a compound to become a priority, it first must have significant health effects and potential for exposure. Then available technology for control of the compound and economic costs are considered. This method can eliminate contaminants from consideration based on technology limitations or economic factors. Compounds that are deemed toxic and have high occurrence, but are not technologically or economically feasible to control, can be identified as candidates for new technology research.

The AWWA screening approach first considers toxicity criteria, which are qualitative and based on existence and quality of data rather than magnitude of effect. Then, the screening process evaluates the potential for exposure. Because it's qualitative, it can prioritize chemicals even in the absence of information.

3.4.4. Harmonized Integrated Hazard Classification System

The Harmonized Integrated Hazard Classification System for Human Health and Environmental Effects of Chemical Substances was created by a meeting of OECD's 28th Joint Meeting of the Chemicals Committee and the Working Party on Chemicals in 1998 [26]. This system was created to provide a simple and transparent way to standardize the classification of chemicals (internationally). Once classified, measures can then be taken to avoid or manage potential risks in circumstances where exposure may occur. For many endpoints in the scheme, the criteria are semi-quantitative or qualitative and expert judgment is required to interpret the data for classification purposes. For some endpoints (e.g., eye irritation) a decision tree approach is used.

Various harmonized classification systems are presented for the following groups of chemicals: chemicals that cause acute toxicity, skin irritation/corrosion, eye irritation/corrosion, respiratory or skin sensitization, mutations in germ cells, cancer, reproductive toxicity, target organ oriented systemic toxicity, and hazards for aquatic environment.

In this system, chemicals are classified by exposure route (oral, dermal, or inhalation) and then are categorically classified by level of exposure/toxicity (up to 5 categories). In the case of aquatic exposure, chemicals are classified by chronic and acute exposure risks. The classification of a chemical substance in this manner depends both on the criteria used and on the reliability of the test methods underpinning the criteria, as well as the quality of the expert judgment utilized. This method allows for categorization of different chemicals based on their potential for human and environmental health impacts but does not include a methodology to rank chemicals within categories or to assess risk of exposure due to occurrence in different environments. More information is available at the following address:

http://www.epa.gov/oppfead1/harmonization/docs/doc/integr~1.doc .

3.4.5. Use Clusters Scoring System (UCSS)

This method identifies and screens clusters of chemicals ("use clusters") that may be substituted for one another to perform a particular task. The U.S. EPA uses risk scores generated by UCSS to prioritize chemicals and clusters for further investigation using human and environmental hazard and exposure data from a number of sources including such sources as the Toxics Release Inventory and IRIS [72]. This method allows a user to enter data indicating the potential for human and ecological exposure and hazard, and, unlike many other models, the level of EPA interest. This database contains data on nearly 400 use clusters and 4,700 chemicals, and uses the Chemical Abstracts Service Registry (CAS) number.

In the next section, a different class of methods that rely on statistical analysis are discussed. These methods are often based on existing decisions. That is, statistical tools are applied to identify the relationships among input variables, and models based on these relationships are created. Therefore, they are referred to as prototype methods.

The main difference for experts between rule based methods and prototype methods for classification lies in which part of the process expert judgment is used. In the prototype method, expert judgment is used in the development of the input data, as well as review of outputs, but the rules are derived by the statistical algorithm from past

decisions (a training set). In rule-based methods, expert judgment is part of the decision process in selecting the rules and weights, which potentially increases subjectivity.

4. Prototype Methods

This review of prototype methods is presented in the context of statistical pattern recognition. Statistical models for pattern recognition develop relationships among input variables (a model) that can be used to predict class-association (output) of a given sample using measurements or other descriptive data (attributes). This classifier is usually "trained" using a training data set where the class association of each data point is known. Prototype methods for classification differ from expert qualitative and quantitative rule-based methods in that the algorithms generated from the prototype methods are based on data-derived classification models. That is, the algorithm derives the relationships among input variables and classification based on a training data set in which presumably correct decisions were made. These relationships form the parameters of a model which then can be used for the classification of new contaminants. In prototypes, the relationship between inputs and classification is determined by statistical analysis rather than by experts, which moves the subjectivity in the classification process to data development rather than rule development. Most prototype methods outlined below have the ability to predict the classification of a given observation, but differ in the ability to communicate the relationship between the classification reached and the input features. Methods considered include machine learning, classical statistical approaches, modern statistical approaches, decision tree methods, Bayesian belief networks, and genetic algorithms.

Classification and pattern recognition have a long and respectable history in engineering, especially for military applications. One of the objectives of pattern recognition is to delegate tasks from human experts to machines. One class, neural networks, has arisen from analogies with models of the way humans might approach pattern recognition tasks, although as machine learning approaches they have developed a long way from their biological roots. Neural network methods have had great impact on pattern recognition practice, including emphasis on the need for families of models with large but not unlimited flexibility given by a large number of parameters in large-scale practical problems. These models can be seen as fitting in between the parametric and non-parametric statistical methods for pattern recognition. A parametric statistical method is based on a family of models assuming a normal distribution with a small number of parameters, and a non-parametric statistical method uses models that are more flexible because they do not require information about the underlying distribution of the data.

All prototypes share the characteristic that their performance is largely derived from the quality of the training data set to represent the data requiring classification. The validity of the final classification depends on choosing appropriate data for training; hence, expert judgment remains a key part of the process.

Below, several prototype alternatives are reviewed, including artificial neural networks, classification and regression trees, support vector machines, and Bayesian belief networks.

4.1. ARTIFICIAL NEURAL NETWORKS (ANN)

An Artificial Neural Network (ANN) is a information processing paradigm conceptually based on the mode of learning of networks of the human nervous system. ANNs apply a highly flexible non-linear function parameterized by unknown weight coefficients and a number of hidden layers. Because the mathematical form of the non-linear function is very complicated, it is difficult to communicate about them with non-specialists. However, because the non-linear function is also highly flexible, ANN can be developed to capture almost any kind of underlying relationship between input and output data. The model developing process includes a decision about the number of hidden layers (the larger the number of hidden layers, the more complicated the model) and estimation of the unknown weight coefficients. Limiting the number of hidden layers reduces the number of combinations and hence the complexity of the mathematical relationship between inputs and outputs. ANN models use a set of interconnected processing elements (neurons) that adapt to classify data through a learning process as the network processes the data. ANNs differ from rule-based methods and classical statistical models in that non-linear relationships between inputs and output are possible. For classification purposes, ANNs apply weighting to variables in non-linear functions and do not specify a functional form (such as quadratic or cubic) as do statistical models.

Statistical parameter estimation is often referred to as the learning process in the neural network terminology. The general idea of a learning process is to find the parameter values (the weights) that minimize the discrepancy between the model prediction and the observation. The discrepancy (E) is measured by the conditional likelihood for a classification problem. Because (E) has a very complicated mathematical form, the minimization process is not trivial. The minimization process involves calculations of partial derivatives with respect to the weights as well as the partial derivatives with respect to the inputs and outputs. The frequently used *back propagation* method carries out the calculation iteratively, including a forward pass to calculate the outputs from the inputs and a backward pass to calculate the derivatives with respect to the weights.

ANNs fall into distinct learning types: supervised, unsupervised, and a hybrid combination of the two. *Supervised learning* is a process of training a neural network by providing samples that contain both input and output information. The neural network determines (learns) the relationship that exists between the inputs and outputs of the training data. Once trained, the neural network can then predict outputs or classification from the input data. This technique is mostly applied to the *feed forward* type of neural networks, where signals travel in one direction, from input to output.

A feed-forward neural network has units that have one-way connections to other units, and the units can be labeled from inputs to outputs. The units can be arranged in layers so that connections go from one layer i to a later layer j. The links have weights, w_{ij}, which multiply the signals traveling along them by that factor. These

units are also known as interconnected processing elements or neurons. A three-layer feed-forward neural network can be represented by the following equation:

$$y_k = f_k\left(\alpha_k + \sum_{j\to k} w_{jk} f_j\left(\alpha_j + \sum_{i\to j} w_{ij} x_i\right)\right) \qquad (1)$$

where x_i are inputs, and f_j is a function. The functions are almost inevitably taken to be linear, logistic or threshold. From a statistical perspective, a feed-forward neural network is a linear or non-linear regression model with the model format decided by the number of layers and the number of units. Ultimately, it is an approximation problem, i.e., equation (1) is used to approximate the underlying "true" classifier.

An *unsupervised learning* network adapts to the data that is provided and finds structure in the data. Unsupervised networks work to cluster and reduce the dimensionality of the data into a smaller number of features. Unsupervised networks learn by analyzing principal components (cooperative) or by competitive (competing) learning. *Kohonen networks*, an unsupervised model, use competitive learning to predict binary features (0 or 1) and produces results very similar to the statistical model of *k-means clustering*, described below. When used for classification purposes, unsupervised networks require labeling of classes after the model is created.

A perceptron is a group of neurons in one layer acting together on the same input information. A perceptron can have one or more outputs and, as the network attempts to predict the correct outcome, each output is assigned a weight proportional to the difference between the desired and actual outcome. A perceptron with a linear activation function (linear relationship) produces results similar to a linear regression model and generally is as easily understood. However, hidden layers of neurons can be added to the ANN to form a *Multilayer Perceptron* (MLP), similar to multi-variate non-linear regression. The output weights for the MLP are not only derived from the interaction with the inputs and the hidden layers of neurons but also derived from the interaction of the hidden layers of neurons to the outputs. This hidden interaction creates a new non-linear variable which can make it difficult to elucidate the weights used to generate the outputs and features influencing the model. Determining the correct number of neurons can be difficult because too few cannot correctly predict the outcome, while too many memorize the data, and thus cannot generalize.

Back Propagation (BP) can be used when training a MLP, and often applies the generalized delta rule. After the initial input of data and prediction, the predicted outcomes can be compared with the calculated outcomes. This output error is calculated at the hidden nodes and is used to alter the weights of the output units. BP occurs for each of the observations in the training data set and is repeated until the error is at its minimum. Because of the number of iterations it takes to minimize the errors and train the network, BP can be very time consuming. There is a danger of over-fitting the data to the point of creating an uninformative model too specifically targeted to the training data set. Thus, some minimal level of error must be tolerated.

Radial Basis Function (RBF) networks do not divide space into hyperplanes as most neural networks do but into hyperspheres characterized by its center and radius.

Hidden neurons compute a function that is generally bell-shaped (Gaussian) and the attributes of this function are derived from unsupervised learning. Supervised learning is applied to the outer layers of neurons, so the RBF is an example of a hybrid neural network. Because the hypersphere function is already not linear, only one hidden neuron layer is necessary to model any shape of function. This makes RBF networks generally quicker to train and more transparent than other MLPs.

The Probabilistic Neural Network (PNN) is designed for classification purposes and estimates probability density functions of different classes of observations. In the PNN, there are at least three layers: input, radial, and output layers. PNN assumes that the proportional representation of classes in the training set matches the actual representation in the population being modeled. An advantage of PNN is that the output is probabilistic which increases its ease of interpretation. However, it is only an advantage if prior probabilities for the classes can be calculated. Training a PNN happens almost instantaneously but a PNN network requires significant computer memory and is slow to execute.

4.1.1 General Applications of Neural Networks

Neural networks learn from experience so they often function well addressing problems that include pattern recognition and forecasting. ANNs can be trained to recognize patterns with information provided in a training data set. Pattern recognition with ANNs has been extremely useful with respect to diagnosing medical problems, detecting credit card fraud, and making identifications. ANNs were found to diagnose heart attacks with greater accuracy than emergency room physicians [1]. Cervical cancer can be diagnosed using ANN technology to examine cell culture information from pap smears[42]. ANNs also have been used to detect credit card fraud by signaling to credit card companies' changes in customer spending habits [50] and in facial recognition software, primarily used as a security measure [82]. When combined with information from Geographic Information Systems (GIS), ANN technology can help to identify patterns from remotely sensed data. GIS data are currently being used to model water quality by using ANN to analyze spatial information and watershed properties of a groundwater system [37].

Neural networks can predict and classify information, and recent work has shown these models to be particularly applicable with respect to prediction of water quality, demand, and treatment. ANNs are being used in Australia to forecast surface water quality [9]. Both the timing and magnitude of toxic algal blooms are being predicted using ANN algorithms [9, 73]. Similarly, ANN modeling is being used to predict concentrations of Cryptosporidium and Giardia using biological, chemical and physical parameters obtained from a water treatment plant [81]. Also, ANN models were found to accurately predict the biodegradation and aqueous solubility of a set of organic chemicals [30,31]. The demand for water has been accurately forecasted using ANNs [24]. Additionally, ANN models were found to be helpful in predicting the dose of coagulant needed to remove organic molecules in water at a water treatment plant and may prove useful in automating the dosing of coagulant [52]. The USGS Surface-water quality and flow Modeling Interest Group (SMIG) has a website summarizing many of the applications of neural networks to environmental systems [28].

4.2. STATISTICAL METHODS

Statistical approaches to classification generally have an underlying probability model, which estimates the probability of each observation being in a certain class rather than simple classification of data. This probability may be used to "rank" the observations by their probability; those with the highest probabilities of being in a class of concern may have higher importance. Statistical methods may be used in basic data analysis but for classification require a specific pre-determined model (e.g., linear regression). However, these models can be used in classification exercises when training data sets are used to determine the class of new observations. Some statistical models are nearly identical to other prototype methods but have underlying structures that are not hidden and may be more easily communicated. Classical statistical models focus on the fit of the data parameters to a particular model, which may not be the best way to classify diverse sets of data, such as drinking water contaminants. However, modern approaches first analyze data to inform model selection, which may be better suited to classification problems.

4.2.1. Discriminant and Regression Functions

Fisher's Linear Discriminant Analysis (LDA) is one of the oldest parametric classification methods. This method divides a data set with a series of lines by drawing discrimination lines that bisect a line joining the middle of two groups of data. This method also can be used with more than 2 dimensions of variables by fitting planes (3 dimensions) and hyperplanes (>3 dimensions). When there are only two classes to predict, the Fisher model is equivalent to a multiple regression problem with attributes predicting numerical classes (0 or 1). Logistic Regression (sometimes called Logistic Discrimination) also uses linear discrimination lines to predict numerical classes (0 or 1) but uses slightly different separation criteria than Fisher's. In practice, both methods produce similar results and the Linear Discriminant provides good starting values for using the Logistic method for classification.

For more complex models or models with more than two classes, a Quadratic Discriminant provides a quadratic discrimination surface. These methods can be used to generate a discriminant function with a training set of pre-classified cases and the function can be used subsequently to predict class membership with new observations. Complex data, in most cases, can be fitted with complex regression models. *Flexible Discriminant Analysis* (FDA) is a generalized form of discriminant analysis where more flexible (i.e. non-parametric) regression can be used [22]. With these analyses, the weighting for different features are more transparent than in neural networks and allow for easy interpretation of the results. However, unlike neural networks, interactions between features need to be specified before the model is run because some features may have a synergistic effect on one another. There are two related forms of discriminant analysis methods: *Penalized Discriminant Analysis* (PDA) and *Mixture Discriminant Analysis* (MDA). FDA can be seen as the linear discriminant analysis (LDA) in an enlarged predictor variable space, the same paradigm used in Support Vector Machines, described below.

When the number of predictor variables is large, enlarging the predictor space is not advised. PDA fits a linear discriminant analysis model but penalizes its

coefficient to be smooth [21]. Loosely speaking, the PDA assigns less weight to "rough" coordinates and more weights to "smooth" ones. The penalty applies to linear combinations that are rough.

Both FDA and PDA are similar to LDA in that they are simple prototype classifiers: a new observation is classified to the class with closest centroid. In many situations a single prototype is not sufficient to represent non-homogeneous classes, and mixture models are more appropriate. MDA expands the simple prototype idea to model each class by a mixture of two or more Gaussian distributions with different centroids [20]. This allows for more complex decision boundaries. MDA can be seen as a generalization of FDA and PDA.

Nonparametric logistic regression is a special case of generalized linear models, in which some of the linear dependencies are relaxed to allow fitting a non-linear curve. Logistic regression estimates the probability of an outcome or classification as a function of predictor variables, derived from a training data set, similar to prototype methods discussed in Section 4.3. In nonparametric logistic regression, a model need not be specified, rather the data determine the form of the function used to predict classification. The linear model is replaced with a (transformed) smoothing function, which can be fit using the roughness penalty approach [14]. A nonparametric logistic regression model could be derived from a training data set by minimizing error, and allowing nonlinear curve fitting.

4.2.2 Clustering Algorithms

Traditional statistical methods for unsupervised classification include *k-Nearest Neighbor* (kNN) and *Cluster Analysis (Clustering Algorithms).* kNN is a non-parametric method; new observations to be classified are compared to a training data set. It assumes that members of the same class in the training set will be clustered in some non-random manner. The distance from the new observations to the closest "k" number (user-selected) of observations in the training set is measured. Distances can be measured in a number of ways and measurements tend to get complex with multi-variate data. The presence of irrelevant or unnecessary variables is problematic with this method.

Cluster analysis is a method by which large sets of input data are grouped into clusters. A clustering algorithm attempts to find natural groups of features within a set of observations based on some similarity. To determine cluster membership, most algorithms evaluate the distance between a point and the cluster centroids. The algorithm produces a statistical description of the cluster centroids. Although there are many variations in how clusters are determined (distance measures, probabilities, etc.), there are two main types of cluster analysis: hierarchical and non-hierarchical. In hierarchical clustering, long chain or nested clusters (subgroups) are created and there are no predefined classes. In non-hierarchical clustering the data are partitioned directly into their final groups. *K-means clustering*, a non-hierarchical method, splits observations between a user-defined k number of clusters. Observations are assigned to groups in different iterations until individual clusters are internally similar and maximally dissimilar from other clusters. If more than one feature can be used to define the clusters, the distances (dissimilarities) between clusters are measured in

multi-dimensional space. As in other statistical methods, training sets can be used to build the clustering algorithm to be used on later observations.

As a method of classification, cluster analysis is unsupervised and often considered exploratory because of a lack of predefined classes. This method presents the danger that it will find clusters in every data set, so an effort must be made to make sure that the clusters are meaningful.

4.3. MODERN STATISTICAL METHODS

Modern statistical methods are mostly computation intensive. In general, they make fewer probabilistic assumptions about the data. Very few methods have been tested for general application, although most have been applied as analytical tools for environmental decisions. We briefly introduce several examples, including: classification and regression tree analysis (CART) - a decision tree model, the support vector machine, and Bayesian belief networks.

4.3.1 Decision Tree Methods

A decision tree is a classification scheme that involves a series of tests, each with a mutually exclusive and exhaustive outcome. This method is most commonly used with categorical data and the classification outcome results from a sequence of logical steps through any number of tests. An individual test within a classification scheme may involve one feature or multiple features (multivariate) and may have two (binary) or more outcomes. The graphical "tree" is derived with a test at each node and with outcomes branching from each node. By branching according to the outcome of each test, one arrives at a "leaf" that contains the classification of an individual observation. These schemes are usually very transparent but can get complex when using multivariate tests. As with other methods, training data can be used to construct a decision tree and then additional observations can be classified using the tree. Decision trees do not have distributional restrictions and are not limited to linear correlations, so they offer advantages over discriminant analysis.

The logic of this method is sometimes employed within the context of expert judgment. Experts ask a series of questions or tests (e.g., is the contaminant toxic? or how toxic is the contaminant?) and with each response, they break down the list into smaller and smaller groups until final satisfactory classifications are reached.

Tree-based modeling is an exploratory technique for uncovering structure in data [2], increasingly used for:

(1) Devising prediction rules that can be rapidly and repeatedly evaluated;
(2) Screening variables;
(3) Assessing the adequacy of linear models; and
(4) Summarizing large multivariate data sets.

Tree-based models are so-called because the primary method of displaying the fit is the form of a binary tree. Advantages of tree-based modeling include: easier interpretation and communication than linear models when the set of predictors contains a mix of numeric variables and factors; invariance to monotone transformation of predictors so that the precise form in which these appear in a model is irrelevant; and

ability to capture non-additive behavior (the standard linear model does not allow interactions between variables unless they are pre-specified and of a particular multiplicative form).

A tree-based method (also known as the classification and regression tree, CART) can be used for uncovering structure in data, identifying prediction rules that can be evaluated, as well as for identifying screening variables, and summarizing multivariate data sets. Models are fit by splitting data into successively smaller and smaller subsets. CART allows interaction among variables, and can be used with both numeric and categorical data. The method is attractive to many exploratory environmental studies due to its capability of handling both continuous and discrete variables, its inherent ability to model interactions among predictors, and its hierarchical structure. Further, CART addresses missing variables by using a surrogate measure based on the best available information, so data are not dropped if there are data gaps.

A CART model is fitted by using the recursive partitioning method, which can be described on a conceptual level as a process of reducing the measure of "impurity" [2]. The impurity is often measured by deviance. A node containing a number of observations is considered "pure" if all the observations have the same response value (numerically equal to each other or belong to the same class). The objective of the recursive partitioning method is to successively split the data set into sub-groups (nodes) and make the resulting nodes as pure as possible. The procedure compares all possible partitions and selects the one that reduces the impurity the most.

Without stopping rules, the recursive partitioning will over-fit the data resulting in a model with one observation in each final node. This is certainly undesirable because an over-fit model will almost certainly perform badly when used to predict future cases. The cross-validation simulation method is a commonly used procedure for selecting the "right" size of a tree, which minimizes a model's prediction error.

Using CART as an exploratory tool, one can greatly reduce the dimensions of the predictor variables to facilitate model development. In addition, CART-selected predictor variables can be used as the screening variables. Using a training data set with a limited number of variables, CART can produce a classification based on the training data, and the classification reported with a misclassification probability. The misclassification probability could help decide whether further information is needed.

A disadvantage of the recursive partitioning method is that the final model may not be the best model. This is because the model fitting method selects each split based on the amount of impurity reduction of the split only. Once a variable is split, it cannot be reversed later. It is possible that a less optimal split at an early stage may result in an overall better model.

CART is described in more detail elsewhere [2, 49]. CART models have been used by EPA to identify predictor variables for modeling fish tissue concentrations of mercury [49]. CART has also been used to: identify predictive factors controlling pesticide levels in a watershed [48]; develop predictor variables for medical diagnosis of asthma [13]; and to identify predictor variables affecting ecosystem dynamics in a brackish lagoon [40].

4.3.2 Bayesian Belief Networks

Bayesian networks stem from the expert decision making process. BayesNet is a graphical model for probabilistic relationships among a set of variables. It differs from a traditional expert system in that a belief network uses a set of algorithms to manipulate the probabilities of class association in an automatic way to present conclusions, such as the posterior probabilities of the various classes. One advantage of the belief network is its ability to take in qualitative knowledge; it behaves more like a human expert. Belief networks are also known as the Bayesian expert systems, Bayes(ian) net(work)s, belief net(work)s, causal (probabilistic) networks, probabilistic expert systems, and probabilistic reasoning on causal graphs. Belief networks are designed and trained to answer more than just the question of classifying future cases. They are able to give a much higher level of explanation, including exploring what were important input features in reaching the conclusion and whether the input data were in some sense in conflict. To do so, they model the whole joint distribution.

For example, suppose we have a set of features $x_1, \ldots, x_m$. We want to find the posterior probability $p(k|x)$ to classify a future case. The rule called naïve or idiot's Bayes takes:

(2)

$$p(k \mid x) \propto \pi_k \prod^{m} p(x_i \mid k)$$

To reach the posterior probabilities, we need a set of qualitative (conditional independence) and quantitative knowledge (prior probabilities of class association). The belief network methods are more complicated than the classification trees. It is often questioned whether the ability to feed in qualitative knowledge actually improves accuracy of classification. Many studies found the answer is equivocal.

Based on expert judgment on the subject, the features or variables $X=\{X_1, \ldots, X_n\}$ in the problem are connected by using a directed acyclic graph (DAG), shown in Exhibit 1. A DAG is a graph with arrows connecting the variables to create a network without cycles. The nodes in a given network (or DAG) are in one-to-one correspondence with the variables. A particular variable X_i may have parents, which are variables with arrows pointing to X_i. The parents are the variables that have a dependence relationship with X_i. The network summarizes the qualitative information about conditional dependence (independence) of the variables. Let Pa_i be the parent of X_i. The joint distribution of X (the quantitative information) is given by:

(3)

$$p(x) = \prod_{i=1}^{n} p(x_i \mid Pa_i)$$

In other words, variable X_i is dependent only on its parents Pa_i. A set of expert assessed prior probabilities is used as the initial starting point of a Bayesian network. These probabilities are the quantitative information about the variable X.

Upon observation of variable X, these probabilities are updated by applying the Bayes' theorem.

As an example, Exhibit 2 shows a hypothetical Bayesian network (a DAG) with four variables. The variable D indicates whether a person has a certain disease (i.e., there are two classes: with the disease D=1, or without the disease D=0). Variables S_1 to S_3 indicate whether symptoms 1 through 3 are present (present=1, absent=0). The three arrows pointing from D to S_1 - S_3 reflect our belief that whether a person is diseased has relationships with the three particular symptoms. The direction of the arrow indicates the causal relationship (i.e., the disease determines the symptoms). The arrow from S_1 to S_2 indicates an additional causal relationship between the two symptoms.

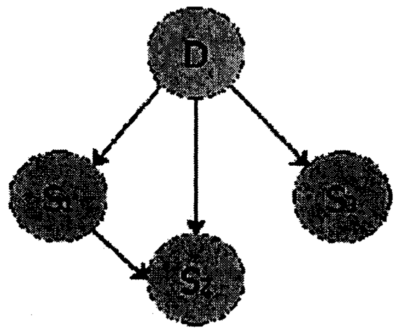


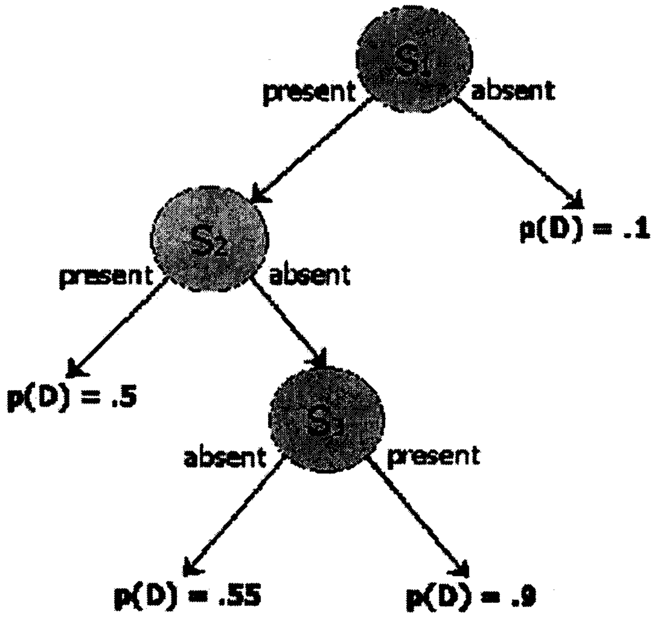
Fig 1. A Hypothetical Bayesian Network

Based on medical information, experts may assess the conditional probabilities associated with the arrows. For example, the arrow from D to S_1 is associated with four probabilities: $p(S_1=1|D=1)$, $p(S_1=0|D=1)$, $p(S_1=1|D=0)$, and $p(S_1=0|D=0)$. With these probabilities, we compute the probability of a patient having the disease after assessing whether s/he has all three or some of the three symptoms ($p(D=1|S)$). This final result is based on mainly expert opinions (the prior probabilities). Once the patient is confirmed with the disease but with only symptom 1, we can enter the data ($D=1$, $S_1=1$, $S_2=0$, $S_3=0$) into the model and calculate all the probabilities in the model including the probability $p(D=1|S)$. The resulting new model is a combination of expert opinions and the data from this patient. Note that the calculation is still possible if only one symptom is known for sure. In other words, incomplete data will not affect this calculation. This feature is important for poorly characterized problems, because there are likely to be data gaps for key variables. The process can be repeated many times. As more information is added, the initial expert judgment of the probabilities will be replaced entirely by the data. However, the qualitative information reflected in the network structure will stay the same, unless efforts are made to search for a structure that better fitted to the data.

In contrast, a typical CART or neural network application only uses the available data. For example, using information from many patients, a CART

application may result in the following model (Exhibit 3). This model stores the proportion of diseased patients with various combinations of present/absent of the three symptoms. In addition, when making future predictions, one needs to collect data on all three symptoms, whereas in Bayesian networks, incomplete data is not a problem.

For classifying drinking water contaminants, the variable of interest would be the class association of the contaminant. This variable would include a vector of five probability values if there are five categories or attributes. These probabilities add up to 1 and provide the basis for classifying the contaminant. In addition, the Bayesian networks provide probabilistic dependence relationships among all variables. These relationships could reveal important insights to information, such as what led to the specific classification decision. Also, these relationships make it possible to handle incomplete data, as long as all variables have values in some observations.



4.3.3

Fig 2. A Hypothetical CART Model

Support Vector Machines (SVM)

A Support Vector Machine (SVM) is a statistical method that can be used for classification or regression analysis but approaches classification in a manner similar to ANN. SVM classifies data in a way that optimizes the differences among groups. The SVM produces a non-linear classification boundary in the original input space. This boundary is a hyperplane determined by certain data points in the training data set, or

the support vectors. This hyperplane is obtained from the solution of a quadratic programming problem.

SVMs are learning machines that map their input vectors into a high dimensional feature space, constructing an optimal separating hyperplane by minimizing a quadratic function with linear constraints. The hyperplane attempts to split the classes of data presented in a training set. The "support vectors" are those observations closest to the boundary of the hyperplane and, therefore, are most useful in generating a solution. The intrinsic shape of the hyperplane is carried out by means of a kernel function. Choosing an appropriate kernel requires statistical knowledge and experience, but some general rules exist to make selection of a kernel easier.

A statistical classification problem divides the input space defined by predictor variables (or features) into a collection of regions labeled according to the classification. Depending on the prediction function, the boundaries can be rough or smooth. Most classical statistical methods are for problems where the decision boundaries are linear. For example, suppose there are K classes, labeled 1, 2, ..., K. The fitted linear model for the ith class is fi(x) = i0 + iTx. The decision boundary between classes i and j is that set of points for which fi(x) = fj(x), that is the set {x: (i0 - j0) + (iT - jT) x = 0}, which is often referred to as a hyperplane. Because the same is true for any pair of classes, the input space is divided into regions of unique classification, with sequential hyper planar decision boundaries. However, this simple linear discriminant has a number of problems [53]. First, when the data are separable by linear hyperplanes, there are many solutions, and the one found depends on the starting values. Second, the computation algorithm may take a long time to converge. Third, when data are not separable, the computing algorithm will not converge. A rather elegant solution to those problems is to add additional constraints to the separating hyperplane, resulting in the optimal separating hyperplane. The optimal separating hyperplane separates the two classes and maximizes the distance to the closest point from ether class. As a result, the separating hyperplane is defined in terms of a linear combination of a few support points. Points well inside their class boundary do not play a big role in shaping the boundary. This optimal separating hyperplane can be generalized to the non-separable case, often known as the support vector classifier.

The support vector classifier finds linear boundaries in the input space. The procedure can be made more flexible by enlarging the input space using basic expansions such as polynomials. Generally linear boundaries in the enlarged space achieve better training-class separation, and translate to nonlinear boundaries in the original space. The Support Vector Machine is an extension of this idea, where the dimension of the enlarged space is allowed to get very large, infinite in some cases [18]. The computational procedure of the SVM employs a feature which balances the roughness of the curve and the classification error. Because perfect separation is typically possible in an enlarged input space, a regularization parameter (similar to the roughness penalty parameter in smoothing) determines how wiggly the boundary can get.

Other than just providing classification information, the SVM algorithm highlights critical observations as support vectors which allows a high degree of data compression, which can help identify areas where additional data may be needed or redundant. Outlying data points can distort the decision boundaries, but emphasizing

methods, such as boosting and active set techniques, can address that problem by increasing the relative statistical weight of critical observations. The final decision boundary generated from SVM is generally non-linear and is considered to be mathematically more reliable than multi-layer ANN, because many ANNs generate boundaries that look non-linear but are in fact joined linear segments. SVM was designed for use with large data sets so a pre-screening method may not be needed with SVM. Although SVM can be used for categorical data, it may be better suited to problems with numerical data.

Improvement of SVM using kernel logistic regression is being proposed [80]. Currently, SVM is more applicable to binary classification problems, however current work is ongoing to address multiple class problems [80]. In addition, SVM suffers from the curse of dimensionality, that is, dimensions are added as more predictor variables are included, increasing data requirements. When the number of predictor variables increases and class separation occurs only in the subspace of few features, SVM can not easily find the structure and is bogged down from having many dimensions to search over. One would have to build the knowledge of the subspace into the model telling it to ignore all but those few inputs. Hastie et al [18] illustrates the curse of dimensionality with an example. When no noise was added to a simulated data set, SVM using a second degree polynomial kernel performed best, but not by much compared to the additive model. However, higher-degree polynomial kernels did much worse. When random noise was added to the simulated data, the SVM performance deteriorated.

4.4. STATISTICAL MODELS VS ARTIFICIAL NEURAL NETWORKS

Artificial Neural Networks (ANN) are created as a form of machine learning by computer scientists and are developed for a specific application, such as pattern recognition or data classification, through a learning process. Statistical methods are created by statisticians to analyze data and develop a model characterizing the data. Strong similarities exist between neural networks and statistical models. Supervised learning ANNs correspond to statistical non-linear discriminant analysis; unsupervised learning ANNs are similar to factor analysis and clustering. Many neural networks now incorporate features of statistical models and can be implemented with standard statistical software. Additionally, more statistical models incorporate features of neural networks, e.g., Support Vector Machines are sometimes considered to be a hybrid statistical model with features of an ANN.

Artificial Neural Networks produce classification models derived from example training data but include "hidden" layers that are difficult to interpret. They are flexible and can approximate any continuous function if the hidden layer is large enough, but the underlying relationships among predictor variables remain unknown. Statistical methods can also provide classification models from training data but tend to be more transparent than ANNs because model assumptions and underlying data distributions are hidden in ANNs. However, some statistical models require prior information about the distribution of the data and expertise in model selection. ANNs determine the relationship between inputs and classification without this input by applying defaults. However, some statistical approaches (e.g., Classification and

Regression Trees (CART) and Bayesian Belief Networks (BayesNet)) apply graphical methods so the relationships of variables can be evaluated.

The standard SVM works well in binary classification, but its appropriate extension to the multiple class case is still an on-going research issue. SVM uses non-linear hyperplanes as the class boundary while ANNs use linear boundaries. In addition, SVM is based on sound mathematical and statistical theories of regression penalty, while the algorithms in neural networks are more or less a black-box.

Decision trees are sometimes viewed with a robustness not shared by the neural networks. Subject matter experts often find trees easier to interpret than any other representation. Decision trees are able to graphically explain the decision-making process, whereas neural networks are not so transparent.

In comparison to CART and ANN, Bayesian Belief Networks have at least three advantages:

(1) Bayesian networks allow learning (model updating) from the causal relationships;
(2) Bayesian networks allow combining prior (or expert) knowledge and data; and
(3) Bayesian networks offer an efficient approach for avoiding over-fitting of data.

Bayesian networks can also handle incomplete data sets, due to the built-in network structure. Although the Bayesian networks are viewed as computationally intensive, classification with a small number of predictor variables (e.g., four or five attributes) may not be too difficult.

5. Model Selection (Model Assessment)

Proper model selection or assessment is essential for any prototype or statistical classification method. Model selection/assessment is the process of selecting the "right" number of predictor variables and the "adequate" complexity of the statistical model. When a model is "over-fit", or over complicated (e.g., too many splits in CART, too many layers in ANN) a model can fit to the training data set well but may perform poorly when applied to an independent data set. An under-fit model will not adequately predict classification. The step-wise model selection process of a linear regression model is an example of model selection. The step-wise procedure fits a series of models each with a different number of predictor variables. The models are evaluated using the Akaike Information Criterion (AIC), which is the sum of model deviance (a measure of model fit) and a penalizing term for model complexity. As more predictor variables are included, the model deviance will decrease, but the complexity is increasing. Also, as the number of predictor variables increases, the incremental decrease in the deviance decreases. When the incremental decrease in model deviance does not compensate the increase in model complexity, the predictor variable should not be added.

Cross-validation is another example of model selection. The idea of a cross-validation model selection procedure is directly related to avoiding over-fit. In a cross-validation, a subset of the training data is randomly chosen to fit the model. The fitted

model is then applied to the remaining data for evaluating predictive error. The process is repeated many times for different model complexity, and the resulting model prediction error is graphed against the model complexity. The resulting line almost always has a minimum point. That is, as the model complexity increases, the predictive error will decrease initially and reach the minimum. Further increases in model complexity will result in increased predictive error. The model complexity corresponding to the minimum predictive error is usually selected as the "right" complexity for the model.

For CART, model assessment is aimed at selecting the right size of the final tree. The cross-validation approach is a well developed (and most frequently used) technique for selecting the tree with least predictive error. In the original CART [2], a cost-complexity measure similar to the AIC was suggested for model selection. However, the cost-complexity measure and AIC tend to over-fit [53]. As a result, cross-validation is considered the standard method for CART model selection. Typical cross-validation results are presented graphically. The model predictive error (measured in deviance) is plotted against the size of the tree model (number of terminal nodes). The number of nodes corresponding to the minimum deviance is usually chosen as the correct model size.

Two things have to be decided for selecting an SVM model. First, because SVM is a penalization method, where the model error is balanced by the roughness of the decision boundaries, one must choose the regularization or smoothing parameter. Second, as in all models, one must select the right predictor variables. In theory, both tasks can be guided by a cross-validation simulation. The idea of a cross-validation for SVM is similar to the cross-validation for CART. When used for selecting the smoothing parameter, the predictive error (measured as the loss, see [18]) is plotted against the smoothing parameter values. A small value corresponds to a rough and wiggly separating hyperplane and a large value corresponds to a smoother plane. Using cross-validation for predictor variable selection has not been seen in the literature.

For a BayesNet model, the objective of model evaluation is to determine the number of connected nodes and competing model formulations. The complexity should be determined by available data and other information. One way to begin is with an exhaustively complex model and eliminate variables based on the data and available qualitative and quantitative information for model fitting. A more interesting problem is how to select from competing models reflecting different theories of causal relationship. Stow and Borsuk [61] presented an example of comparing two competing theories of causal relationship between Neuse River estuarine fishkills and the presence of *Pfiesteria piscicida*, a microorganism which creates a toxin harmful to fish and humans. In the paper, one model represents the theory that toxic *Pfiesteria* causes a fishkill and the other model represents the theory that a fishkill simulates the formation of toxic *Pfiesteria* from *Pfiesteria*-like organisms. By comparing the estimated probabilities of all the causal relationships defined in both models, one can judge which model is more likely. This approach is appealing since we can now judging a model's relevancy not only by the model error statistics such as residual sum of squares and predictive deviance, but also by other independent subject matter information. The model selection process for BayesNet is inevitably iterative and would requires interaction between model developers and subject matter experts.

Model selection for neural networks requires addressing several issues. For example, the selection of starting values for the weights. Using zero as the starting point, the algorithm will not move. Starting with large weights may lead to poor solution. In addition, because neural networks have too many weights and will over-fit the data, many neural networks algorithms have an early stopping rule to avoid over-fitting. As a result, the model is only trained for a while and stopped before the global minimum is achieved. Since the weights start at a highly regularized (often linear) solution, this has the effect of shrinking the final model towards a linear model. Furthermore, the error function (or the discrepancy) is non-convex, possessing many local minima. Thus, the final solution obtained is dependent on the choice of starting weights. One must at least try a number of random starting configurations, and choose the solution giving the lowest error. Ripley [53] suggested using the average predictions over the collection of networks as the final prediction. Hastie, et al. [18] suggested that the ANN model fitting be treated as a roughness penalty problem. In other words, an explicit penalty term can be added to the ANN model error function. This penalty term has a smoothing parameter and is a function of weights. The method is called weight decay. Selecting the right smoothing parameter will lead to shrinking non-essential weights closer to zero. Conceptually a cross-validation procedure can be used for selecting this smoothing parameter. Once the smoothing parameter is selected, selecting the number of hidden layers may not be necessary since non-essential weights are now close to sero. No literature was identified on proper model selection in terms of choosing the right predictor variables for ANN.

5.1. SUMMARY

In summary, a range of qualitative and quantitative approaches to classification in environmental decision making are available. For prototype methods, ANNs have been applied to numerous environmental problems, however their lack of transparency is of major concern for statisticians as well as for others impacted by the predictions. Other statistical approaches that use graphical techniques, e.g. CART and BayesNet, could be used as more transparent and perhaps more informative prototype classification algorithms. Because of the importance of model selection, it may prove essential to test methods side-by-side with a training data set and cross validation procedures.

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Attachment A. Additional Rule Based Methods

A.1. Proposed Regulation Development Process (RDP)

The Proposed Regulation Development Process was developed by the AWWA together with the National Association of Water Companies, Association of Metropolitan Water Agencies, and the Association of State Drinking Water Agencies to help determine if the contaminants found in ambient water should be regulated. With this method, added weight is given to occurrence data that show a contaminant has a wide geographic distribution or special health concerns. Exposure is then estimated using information on the frequency of occurrence from the National Contaminant Occurrence Database combined with information on the population potentially exposed to the contaminant. Toxicity information is provided as a Maximum Contaminant Level Goal (MCLG), or, in the case of carcinogens, as a dose response curve. If no toxicity information is available for a contaminant, the contaminant will not be evaluated. Unlike the AWWA Screening process, this methodology does not eliminate a contaminant for consideration from the list because of economic factors.

A.2. Interagency Testing Committee Approach (ITC)

This approach was devised to screen and recommend chemicals (and chemical groups) for potential rule making by EPA with demonstrated adverse health or ecological effects that have a risk of exposure. The ITC has used three selection processes and the current process (since 1989) uses a computer program to evaluate thousands of chemicals and provide new estimates as additional data become available.

Exposure is determined by matching the Chemical Abstract Service (CAS) number with CAS numbers in a variety of exposure databases. A team of experts determines which chemical substructures are likely to be toxic and then matches them with data from the Toxic Substances Control Act. A computer integrates exposure and toxicity information and the results are reviewed at a workshop where chemicals are selected based on consensus. This allows a large number of chemicals to be considered but also utilizes expert judgment.

A.3. California Safe Drinking Water and Toxic Enforcement Act of 1986

With this method, an expert panel creates a list of chemicals known to cause cancer or developmental and reproductive toxic effects. A tracking database of chemical candidates was created based on information from state agencies and literature searches. Chemicals are categorized on the list based on toxicity information. Toxicity criteria are qualitative and based on toxicity endpoints. Chemical candidates are then addressed by the panel in order of priority based on exposure information. Exposure risk is qualitatively determined to be high, medium, low, no identified concern, or inadequate data. A draft report is released for public and scientific comment before the final hazard priorities are set.

A.4. Hazard Ranking System (HRS)

This is the primary method in which the EPA ranks hazardous waste sites and places them on the National Priorities List to be cleaned up under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Each site is ranked based on the CERCLA hazardous substances list. HRS evaluates the following at each site: Likelihood of Release (LR) into groundwater, surface water, soil and air (exposure), Waste Characteristics (WC) which scores the quantity of the contaminant as well as its toxicity, mobility, persistence, and bioaccumulation potential, and the Target factor (T) for each pathway (ground water, surface water, soil and air) for people and sensitive environments affected by the release. A hazard score is obtained for each pathway using LR, WC, and T scores. After a score is calculated for each pathway, all pathway scores are combined using a root-mean-square equation to determine the overall site score.

A.5. CERCLA Priority List of Hazardous Substances

CERCLA requires that substances hazardous to human health be ranked and placed on the National Priorities List. The HAZDAT database contains occurrence, exposure, and health effects information for 2800 chemicals. If a contaminant is listed in the HAZDAT database and is found at three sites, it will be placed on the priority list. A contaminant's total score is based on its frequency of occurrence, toxicity, and potential for human exposure.

A.6. Pesticide Leaching Potential (PLP)

This method from the Office of Pesticide Programs determines the annual risk of groundwater contamination from pesticide use. This method has only been applied to pesticides used on potatoes and apples. It applies pesticide mobility information for soil and groundwater depth that is not typical of most of the United States. Priority status is given to those chemicals that do not attenuate quickly in the environment.

A.7. Source Ranking Database (SRD)

This methodology was developed for the EPA Office of Pollution Prevention and Toxics, Economics, Exposure, and Technology Division, Exposure Assessment Branch (EAB). SRD performs a screening level review of over 12,000 potential indoor pollution sources to identify high priority pollutants or categories of pollutants for further evaluation. This method produces risk-based rankings by combining the estimated indoor-air concentration (exposure) with a predetermined hazard score for a particular chemical in a particular environment. Users of this system can select product groupings and can categorize exposure as "high" or "average".

A.8. Screening Level Tools

The EPA Office of Pollution Prevention and Toxics has created several Screening Level Tools that are designed to quickly "bin" chemicals by priority for future work. They were developed to be easy to use, fast and conservative. These tools are often used in the absence of monitoring data or to compliment exposure related data. Screening Level Tools include the Chemical Screening Tool for Exposures and Environmental Releases (ChemSTEER), Exposure, Fate Assessment Screening Tool (E-FAST), and ReachScan. These tools require minimal data entry, quickly screen exposure concerns, and create conservative estimates of exposure. These models are included within EPA's Pollution Prevention Assessment Framework.

A.9. Chemical Hazard Evaluation for Management Strategies

Chemical Hazard Evaluation for Management Strategies is a screening level method for scoring and ranking chemicals by potential human health and environmental impacts. It was created by EPA's Risk Reduction Engineering Lab in Cincinnati in 1994. It ranks chemicals and sets priorities for assessment of safer substitutes for major uses by combining toxicity information with exposure information to determine relative risk. An algorithm has been developed to combine and weight evaluation criteria to provide a tool that ranks chemicals according to their potential human health effects (acute and chronic) and environmental effects (aquatic and terrestrial), and their potential for persistence and bioaccumulation.

A.10. The European Union System for Evaluation of Substances (EUSES)

The European Union System for Evaluation of Substances (EUSES) was developed as a method of quantitative assessment of the risks posed by new and existing chemical substances from all pathways to humans and the environment. The system was designed to aid risk managers in making decisions with respect to regulatory actions and is available as a user-friendly computer program. EUSES can be used as an initial screening level tool, or with additional data, can be used for a more refined assessment. Risk assessment is carried out using the following criteria:

- Exposure assessment (estimation of concentration/doses to which humans/environments may be exposed;
- Effects assessment (potential adverse effects identification and the dose-response assessment); and
- Risk characterization (estimation of the severity and incidence of the adverse effects).

Users must have a sufficient degree of expertise to refine the screening level assessment. This method addresses data gaps with estimated data, or default values.

UNCERTAINTY AS A RESOURCE IN RISK COMPARISONS

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Abstract

A pathological debate has evolved on the appropriate role of risk analysis at the federal level in the US. On one hand, substantial academic, public and private sector efforts have developed techniques and justifications for incorporating risk analytical information into "risk rationalizing" decisions. At the same time, a normative critique has jelled around the inadequacy of risk analysis methods to fully describe, and thus to compare, risks (the "holistic" complaint) and the exclusive nature of the risk assessment process (the "anti-democratic" complaint). The past decade has also seen another substantial trend in risk analysis research: improved understanding, description and management of uncertainty. Unfortunately, inadequate attention has been given to merging the normative and technical trends. This has led to several undesirable consequences in the US, Europe, and other developed countries, consequences that include the potential for systematically arbitrary decisions, undermined credibility of risk analysis as a decision input, and pathological debate about the appropriate role of risk information in the risk regulatory debate. This suggests some lessons for developing countries as they adopt risk analytical methods, and undertake risk comparison exercises. In particular, careful attention to uncertainties and the technical debate may provide an opportunity to broach the current normative stalemate in countries that rely extensively on risk data, and to avoid that stalemate in developing countries.

1. Introduction

Debate on the appropriate use of risk analysis for regulatory decisions in some developed counties focuses around the normative appropriateness of various decision rules, especially those that compare and prioritize diverse risks. This debate remains unresolved, and is currently pathological in the sense that proponents of different approaches are not engaged such a way that they can effectively discuss key issues. A more complete, explicit and nuanced understanding of the uncertainties inherent in any risk comparison may provide an opportunity to overcome some obstructions to a healthy debate, with lessons for developing countries and regions that may undertake risk comparison exercises.

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1.1. THE NORMATIVE PATHOLOGY

A prevailing sentiment in the US argues for insulating independent risk analysts from public scrutiny and political manipulation in much the same way that the Federal Reserve Bank provides a level of independence for its economic analysts. This approach is described and argued in detail in *Breaking the Vicious Circle*, a tract produced by Steven Breyer [1] shortly before he joined the Supreme Court. The philosophy behind this approach holds that 1) risk reduction is an appropriate task for the Federal government, 2) risk reduction is implicit or explicit in many regulations, but to date has been pursued in an *ad hoc* and inefficient fashion; 3) risk assessment requires specialized training, and is beyond the capabilities of most people; 4) risk management is in danger of being corrupted to the extent that risk assessors must answer to political or public pressures; and 5) (often implicitly), risk analysis is sufficiently well developed to provide deterministic inputs to risk decisions. Other advocates of risk analysis envision a less exclusive and deterministic role for risk analysis, but there are many within government and without who feel that more reliance on risk as an input into efficient, rational risk reduction would substantially benefit regulatory policy.

A similar concern can be found in Europe, especially in the uncomfortable wedding of the precautionary principle with risk analytical methods. Most recently, in the context of European attitudes towards Genetically Modified (GM) crops, Durodié [2] decries what he sees as pandering to an ignorant and irrational public. Without experts who can make risk judgments, he forecasts "devastating consequences" that will result from "public policy based upon appearances [which are] little more than bigotry." Inadequately restricted public involvement, he concludes, may in fact lead to public injury, as uninformed worry leads to increased psychosomatic illness. If valid for all societies, Durodié's concern bodes ill for developing countries, with rapidly shifting industrialization and urbanization, that do not take immediate efforts to concentrate and insulate expertise. Indeed, the recent SARS outbreak in China and prevalence of AIDS in rural parts of that country would appear to support such an independent infrastructure.

The role of risk analysis as a comparison and decision tool, much less as an insulated specialty, is by no means universally accepted. Critiques come from two directions. The first of these, the "normative critique", challenges the appropriateness of risk analysis as a decision rule component. One version of the normative critique takes the form of an anti-democratic complaint. The concern in this case is that the preferences of risk assessors, which may or may not be consistent with the preferences of the society they are working for, will become institutionalized in risk decisions [3]. Any risk analysis requires a range of assumptions, assumptions that may not have "best" estimates. In such cases the insulation of risk experts, rather than depoliticizing risk regulation, instead enshrines a particular, and limited, set of preferences.

The normative critique also encompasses a holism argument. Here, the concern is that risk assessment is insufficient to capture everything, or even enough, that is important about a regulatory policy. At one level, this means that there may be aspects of a risk (e.g. second order economic effects, violations of moral or religious norms). More pernicious is the concern that risk regulation will come to define risk

preferences, an effect that Shrader-Frechette [4] refers to as "Gresham's Law." In such cases those aspects of risk that can be quantified become important, while those that cannot be quantified are marginalized or ignored. A final dominant concern about risk rationalization regulation is the tradeoff between equity and efficiency, which is part of the broader concern across all economic policy.

Cultural differences provide a powerful, albeit contested, explanation for this difference in risk attitudes between experts and non-experts. Expertise in risk is typically, although certainly not universally associated, for example, with affluence and college education. To the extent that insulation is problematic in the US, Europe and Japan, where college education is relatively common and local, it may further exacerbate the incorporation of useful technical information on novel risks in developing countries. Thus there is a real threat that, without careful planning, the pathology we find in the US and elsewhere will arise even more intractably in countries that begin to adopt technically based risk comparison rules.

1.2. THE TECHNICAL PARADOX

Less attention has been given to a second critique of risk comparisons, that of technical adequacy. Rather than ask whether we *should* follow a particular rule, the technical critique asks whether we *could* do so, even if we all agreed that we should. The technical limitations of risk analysis and the inevitability of uncertainty may in fact be a more fundamental limitation on effective risk comparison than are normative differences. In many cases, even the best risk analysis methods produce findings that are too uncertain to permit clear decisions. In these cases, both advocates and opponents of risk-based decisions should agree that other decision criteria must be sought; risk analysis cannot meet the technical needs of advocates, and in any case is rejected by opponents.

Unfortunately, ideological preference for particular risk comparison rules may serve as a blinder to the technical limitations of risk analysis. Further, this blinder may in the long run undermine the potential of risk analysis to inform decisions. Three unwanted outcomes could result from excessive reliance on risk analysis. First, if advocates of risk analysis in the context of deterministic rules win in political circles, we might unwittingly institutionalize rules that can generate systematic but nonetheless arbitrary decisions. Second, by permitting risk experts broad decision authority we could create conditions that undermine their credibility. Third, we may continue to worsen (in the case of developed countries) or foster (in the case of developing countries) pathological debate about the normative critique. More careful attention to the limitations of risk analysis could not only avoid these three traps, but lead to improvements in these three areas.

With a few notable exceptions (e.g. [5], [6]), much of the policy-related debate in the literature takes one of the two sides in the normative debate. Many proponents of risk analysis work from the premise that efficient reductions of human health risk are the underlying purpose of regulatory interventions [7]. Sunstein [8], for example, works from the assumption that many regulations are inefficient to suggest ways to overcome irrationality, the root cause of that inefficiency. Sunstein [8] and fellow risk rationalization proponents are concerned with the heavy-handed effect of "human

health conservative" assumptions[1], and, like Durodié [2], with the irrationality and ignorance of individuals. A number of studies have been undertaken to demonstrate how inefficient regulations can be (e.g. [9]).

In contrast Montague [10] rejects risk assessment as at best irrelevant (the holistic complaint taken to an extreme) and at worst corrupt (likewise, an extreme version of the antidemocratic complaint. Similarly, Heinzerling [11], [12] takes aim at Breyer [1], and the main study [13] used by Graham [14] in his promotion of risk rationalization. Heinzerling uses unambiguous terms, accusing Graham, and thereby his advocates in the Bush administration and the US Congress, of manipulating data on inefficiency. In particular, she notes that many of the purportedly inefficient regulations have either been eliminated or in fact were never enacted. Where risk rationalization proponents worry about the "heavy hand" of conservative assumptions, opponents counter that risk assessment as currently practiced is so heavily biased towards business interests that conservatism within the analysis plays at most a minor role.

Troublingly, the two sides in this debate are not truly engaged. Neither of the sides addresses the other's central concerns, leading to a debate that is pathological. It won't go away, yet cannot be resolved so long as the issues addressed by the two sides do not overlap. For example, Hammitt's [15] recent and cogent discussion on the relative advantages and shortcomings of valuing life savings using QALYs (quality adjusted life-years) and WTP (willingness to pay) is irrelevant to the debate about whether to use either. If the normative debate is pathological, perhaps analytical issues can provide an avenue for discussion, and thereby a more salutary debate on the role of risk information in regulatory decision making. Further, there is no reason to expect that this will not be transferred to new arenas where risk methods are under consideration.

2. Uncertainty as information for single metric risk comparisons

Elsewhere [16] I have noted how inattention to uncertainty can lead to spuriously precise estimates. In particular, I argue that whether or not one *prefers* to prioritize regulations on the basis of calculated risk/benefit tradeoffs, information may only infrequently permit such prioritizations in practice. When information fails to support decision rules, those rules must be flexible enough to account for uncertainty. I conclude that if the rules are inflexible, and if a bureaucracy is nonetheless expected to follow them, the bureaucracy will do so, generating decisions that are simultaneously systematic and arbitrary.

Systematically arbitrary decisions are no more acceptable to proponents of risk rationalization than they are to opponents. This suggests common ground on which to base improved discussion of the role of risk in decisions. First, while point estimation is useful for shaping the process of risk assessment, point estimates are not appropriate for

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[1] "Human Health Conservative" Assumptions are those that assume a worst or severe case when there is uncertainty. For example, when extrapolating from data on adverse effects to animals to expected effects on humans, if there are two inconsistent data sets, the worse of the two will be selected rather than the best or some average.

comparisons. Instead, risks should be described in terms of a broad list of uncertainties, with quantitative descriptions of plausible values and qualitative description of non-quantifiable uncertainties. A more robust description of risks will appeal to those concerned with the holistic nature of risk, and will at the same time assure rationalists that rankings are not arbitrary.

Thompson *et al* [17] take some steps in this direction by arguing for validation of cost-effectiveness estimates, although the uncertainty bounds generated in their estimates appear unduly narrow, and their decision to report up to seven significant figures appears unwarranted. To successfully address normative concerns, validation needs to address not only costs, benefits and risk estimates, but also sources of information and procedural appropriateness [18].

A second concern is that, given the uncertainties, insulated risk analysts actually threaten rationalist rules. Thus understanding the extent to which uncertainty can lead to universally objectionable decisions argues for increased transparency in the risk assessment process. This will satisfy procedural preferences of those concerned with the anti-democratic complaint, and will ensure that risk analysts are not making unsupportable distinctions between risks. This idea is reinforced by Beierle [19], who found that public participation in risk decisions does not necessary impede, and indeed may enhance, the use of science.

Proponents of efficiency often see risk analysis primarily as an input to decisions. Many advocates of stringent environmental, health and safety regulations see risk analysis as a threat to fairness and equity. Both of these attitudes arise from misunderstanding the primary value of risk analysis: as a tool for understanding complex systems, not as a decision tool. Reduced reliance on risk analysis for point estimates, coupled to broader conception and propagation of uncertainty, may provide the basis for improved debate on risk-reducing regulatory policies. The normative issues, that is, the important but distorted differences in opinions about how things *should be* can be usefully and formally addressed as "normative uncertainty."

3. Uncertainty and Scoping: the New Jersey Comparative Risk Project

Recently Kelly *et al* [20] observed that the European directive mandating careful modeling of the residues of veterinary medicines will not be effective if uncertainty and variability are ignored. As a novel risk within the context of an established risk regulatory regime (the European Union), their finding indicates that exploring possible outcomes may provide a better decision framework than would deterministic, but highly improbably, point estimates. The lesson for developing countries, where input data can be significantly less certain, is clear: exploring uncertainty can avoid surprise, and clarify further research. This in turn favors novel approaches for diverse interests who seek to deal with diverse risks.

Since about 1990, numerous countries and regions, as well as more than half of the individual States in the US have undertaken "Comparative Risk" projects (for details see a number of other chapters in this volume, including Andrews and Linkov). Amongst the most recent (and perhaps at the end of a trend in the US) is the New Jersey Comparative Risk Project (NJCRP), which likewise is explored in some detail

elsewhere in this volume. The Steering Committee charged with overseeing this project faced massive uncertainty associated with aggregation of diverse risks into a manageable set of definitions. This was further stressed by a (self-aware) need to maintain transparency and replicability.

The Steering Committee opted to manage some of the uncertainty in aggregation through a) propagation and b) scoping. Propagation of uncertainty was managed through a novel application of Monte Carlo analysis. Rather than request point estimates of for a variety of cases, the Steering Committee asked the technical analysts to describe several uncertainties around categories in distributional terms, and use Monte Carlo analysis to propagate these distributions through the risk aggregation calculations. This allowed the steering committee members to individually and as a group to weight uncertainty, and use this to contemplate expected values, best cases and worst cases.

A second innovation was the Steering Committee's decision to use scoping methods to explore risk comparisons. They ordered a set of risks under a range of different weighting schemes (e.g. emphasis on: uncertainty, risks to ecosystems, time frame, economic costs), and then evaluated whether there was clear dominance or subordinance under these conditions. Remarkably, a number of risks appeared clearly dominant across weightings, which generated a greater sense of legitimacy among Steering Committee members. Thus, where expected values might have led to normative disagreement about values, scoping permitted an avenue to exploring when this normative disagreement was relevant. In effect, it provided an exit from the pathological dissent...at least for this (diverse) group in this context.

4. Replicable in Other Arenas: Prioritizing Vulnerability in FEMA Mandated Assessments

Among the most significant trends in risk comparison worldwide (although not typically identified as such) is the need to assess vulnerability to natural and human-induced disasters. A number of such exercises are currently underway in developed countries (as well as in the private insurance markets), and surely are in the future of many developing countries. Extremely expensive natural disaster losses in the US, Europe and Japan, as well as increased terrorism concerns, have driven institutions to evaluate, prioritize and mitigate these vulnerabilities.

In the US, the Federal Emergency Management Agency now requires regional vulnerability assessments as a prerequisite for future disaster relief. While the credibility of the threat to withhold relief remains to be tested, vulnerability assessments are taking place across the country. These cases are typified by

- high uncertainty
- low probability / high consequence
- diverse jurisdictions
- public and private risks
- public and private information sources

One approach, which is being tested by the Clark County, Nevada in its GIS-based vulnerability assessment, is sequential aggregation and scoping. Clark County houses Las Vegas, and thus most of Nevada's permanent and itinerant population, as well as Hoover Dam. Due to extreme desert conditions, proximity to faults, dense population and possible American symbolism, flooding, wildfire, drought, disease outbreak, power failure, and terrorism are all potential vulnerabilities. Rather than try to identify the expected risk of every possible vulnerability, and the County decided to follow a modification of the NJCRP approach.

The first step undertaken was to plot historic disasters in a GIS setting, and overlay an exhaustive list of potential future vulnerabilities. From this a series of "possible losses" could be derived, in steps up to and exceeding FEMA relief thresholds. This process (still in progress) will permit possible vulnerabilities to be assessed while maintaining proprietary and sensitive information. It will permit identification of clear non-vulnerabilities, and will delimit further information needs. As Clark County progresses from vulnerability assessment, scoping and uncertainty propagations should remain central to effective vulnerability abatement. The effectiveness of the various vulnerability assessments (dozens are now taking place around the country) should be monitored as information for future comparative risk projects.

5. Conclusion

Beierle [19] finds that public involvement can improve both technical quality and political legitimacy of risk comparison exercises. Likewise, Busenberg [21] finds that when analysis is collaborative—that is, a group of stakeholders decides what to analyze and who will do the analysis—technical information is simultaneously more likely to be accepted and likely to be of higher quality. A credible explanation of these observations is that open and transparent information eases the normative rhetoric. In effect, good information focuses normative debate on appropriately normative differences. As developed and developing countries alike adopt risk comparison exercises, the role of informational uncertainty and collaborative analysis as a tool for developing consensus should not be overlooked.

Risk analysis represents a powerful set of descriptive and exploratory tools, tools that clearly stand to benefit developing countries. Among its appealing characteristics is its capacity to propagate uncertainty. As risk comparison is adopted, whether in the context of narrowly focused, single metric rules or broadly scoped vulnerability assessments, we can expect both improved data quality and legitimacy when uncertainty is viewed as a resource. We will be similarly well served when risk analysis is decoupled from its awkward role as a deterministic input for risk comparisons, and instead is used to explore robustness of decisions as viewed through a range of normative lenses.

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INCORPORATING HABITAT CHARACTERIZATION INTO RISK-TRACE SOFTWARE FOR SPATIALLY EXPLICIT EXPOSURE ASSESSMENT

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Abstract

Site specific ecological risk assessments (EcoRAs) can be improved in terms of technical relevance and managerial utility through the use of spatially-explicit exposure assessment. Formalized descriptions of landscape features (e.g., vegetation cover and physical components of an area) have been used to relate landscape features to the quality of habitat for particular wildlife species. Animals adjust foraging routes and alter daily use patterns in relation to spatial patterns within their home range. The quality of the habitat therefore influences a continuum of wildlife responses including presence-absence, carrying capacity, and dietary exposure to environmental constituents. This chapter describes an approach and a software prototype for combining expressions of habitat quality into spatially explicit risk assessment of contaminated terrestrial ecosystems. The approach and the software are intended for use as a part of a risk-based decision protocol to support the assessment of ecological value and site reuse options.

1. Introduction

Industrial activities create both acute and chronic disturbances in ecosystems surrounding industrial facilities and infrastructure. In the case of the military and some industries, facilities have frequently been inaccessible to the public. As a result, many of these sites are actually relatively undisturbed ecologically, and harbor high biodiversity and large expanses of habitat. Many countries, including developing countries, face the enormous challenge of planning the reincorporation of these sites into the local ecological, economic, and cultural fabric while assuring their safe reuse for civilian, industrial, and ecological purposes. Another challenge is to conduct limited ongoing

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military activities in active sites or industrial operation in a manner having minimal impact on the environment.

In all cases, activities to remediate affected sites must result in the protection of biodiversity, the reduction of present and future pollution, and the restoration of habitats in surrounding ecosystems. These actions will be effective only with an integrated site management approach, which will further support economic development in a manner that is sensitive to the parallel goal of natural resources conservation. In order to accomplish these often dichotomous goals, management specialists and relevant institutions would benefit from a guiding framework that would lead them through a systematic process for planning and decision-making, explicitly integrating both remedial investigation and ecological restoration goals, while considering the socio-economic context.

Ecological Risk Assessments (EcoRAs) are structured to predict the potential effects of stressors (to date, typically chemical) on valued ecological resources. Though much effort goes into evaluating toxicity of chemicals released to the environment, relatively little focus has been directed at the exposure component of the risk equation, and even less attention has been directed toward biological or physical conditions. Consequently, ecological risk assessments often miss major ecological factors that influence the status of valued wildlife species populations. EcoRAs would become more useful in the management decision process if greater attention were given to species-specific site characterization of habitat conditions. This is becoming increasingly more important as criticisms of environmental management are raised, particularly because adverse effects on wildlife populations are not limited to chemical effects. Modification of landscapes, whether for remediation purposes or other landuse practices, can have major consequences for wildlife.

Field naturalists and wildlife managers have understood, at least in qualitative terms, the importance of critical habitat for various life history stages (e.g., nesting sites, winter range, etc.). Animals are drawn to suitable physical structure and food availability, while preferentially avoiding areas of lower quality. The term habitat, though often used loosely as an indication of environmental quality, refers to the combination of physical and biological features preferred by a particular species. What is great habitat for prairie chicken is unacceptable for barred owls. Different habitat preferences reflect evolution and adaptation of species separating from each other in "n-dimensional niche space" (Whittaker 1975). Animals are drawn to particular features of the landscape fulfill their basic life history needs of feeding, breeding, nesting or resting. There are differential area use rates by different species for the same area or the same species in the same area at different times of the year.

Our previous study (Linkov et al. 2001) presents a framework that integrates a number of risk and habitat assessment techniques into a systematic protocol for assessing and managing natural ecosystems at military sites. By integrating proven methods and principles of ecological impact assessment, risk assessment, habitat evaluation, and habitat restoration, the protocol is designed to help managers develop creative solutions to the problem of cumulative stresses to the ecosystem from continuing and past military activities. Linkov et al. (2002), present a model that incorporates spatial scales into exposure assessment and risk characterization for a hypothetical aquatic site.

This paper presents both the methodological approach and a software prototype for spatially explicit risk exposure assessment of contaminated ecosystems. Currently, exposure estimates and subsequent human health and ecological risk projections usually assume a static and continuous exposure of an ecological receptor to a contaminant represented by some descriptive statistic, such as the mean or maximum concentration. These assumptions are generally overly conservative and ignore some of the major advantages offered by advanced risk assessment techniques, such as the ability to account for site-specific conditions and to conduct iterative analyses. The results of this study show that a simple model could explain the contaminant accumulation in ecological receptors foraging in heterogeneously contaminated sites with patchy landscapes.

2. Habitat Suitability Index Models

Vertical and horizontal structure of landscape features (e.g., vegetation, streambed, and soil/rock substrate) defines habitat quality for wildlife. Formal analyses of such features are embodied in the discipline of Landscape Ecology (Turner, et al., 2001). Management of landscape features can be employed to enhance certain species or discourage others (Dale and Haeuber, 2001). Two underlying ecological principles are:

1. increasing vertical and horizontal diversity within a particular area provides a greater number of "niche opportunities" and, hence, more species are likely for the area; and
2. for a given species, a landscape providing preferred habitat quality will support larger, sustainable populations

Recent research efforts have made progress in defining critical relationships among landscape patterns (Roberts & Betz, 1999), scaling issues (Peterson & Parker, 1998), and behavioral response to variable habitat quality. Ejrnæs et al. (2002) used ordination techniques and neural network systems to test classification methods that assigned levels of "naturalness" of vegetation based on simple measures of plant community composition. This classification process enabled more efficient estimates of rarity, nativeness, and uniqueness, than would be required if detailed surveys of rare species, or exhaustive methods to document rare species were undertaken. Freckleton and Watkinson (2002) reviewed literature on regional assemblages of plants and patchiness of plant populations. Their analyses of metapopulation dynamics of plant assemblages corroborated and updated the long-held concept of dynamic mosaic patterns of plant communities existing within ecoregional landscapes. For conservation management purposes, the underlying ecological principles governing patch dynamics suggest that landuse management practices can be used to create desired landscape configurations to manage wildlife populations (both to enhance desired and diminish undesired species numbers).

Habitat Suitability Index (HSI) models have been developed for many species of management interest. Methods to characterize habitat for certain species was formalized by the U.S. Fish and Wildlife Service in the 1990s (Schroeder and Haire, 1993). Currently, there are more than 160 HSI models published, though usage is limited for quantitative predictions of population densities (Terrell and Carpenter, 1997).

The typical HSI model considers a few easily quantified environmental features (e.g., percentage canopy cover, height of understory vegetation, distance to water, distance to permanent human activity, etc.) to parameterize linear models. The environmental features are scaled between 0.0 and 1.0, representing unsuitable to ideal conditions, respectively. Some models are based on rigorous analyses of conditions across a range of population densities and have been field-tested, whereas others represent the best professional judgment of experts for a particular species. Each of the models provides detailed descriptions of the relationships of the species or group of species and the critical landscape features that define the quality of the habitat. Most of the model features were identified by a panel of experts who worked to arrive at consensus descriptions of the landscape attributes in relation to species success.

Storch (2002) provided strong evidence that proper scaling of landscape patches used to calculate HSI values can greatly improve the predictive capacity of the HSI in terms of grouse populations. He used input data from generalized vegetation measurements without regard to spatial configuration. For, example, measures of elevation, steepness of slope, successional stage, canopy cover, occurrence of gaps, etc. were averaged over the different areas (5 ha, 36 ha, 100 ha, 400 ha, or 2,000 ha). This is an extremely useful observation because it greatly simplifies the task of characterizing relevant landscape features used to parameterize HSI models. Moreover, it provides a wider range of possibilities for land management prescriptions intended to influence wildlife populations. Indeed, Morales and Ellner (2002) concluded that to predict individual animal movement patterns, the challenge is now with characterizing individual behavior patterns more than the spatial structure of the landscape. These recent works underscore the value of relatively simple techniques to characterize landscape structure in defining habitat quality for a variety of wildlife species. They indicate that active management of landscape features can have considerable flexibility in terms of precise configuration of features, but that critical attributes must be organized within the appropriate area to be effective.

3. HSI Model Database

We have located 62 Habitat Suitability Index (HSI) models for bird species, 17 for mammals, and 6 for reptiles/amphibians that occupy terrestrial and wetland areas in North America. Each HSI model that has been published includes a map of those areas of the species' range for which the model is applicable. Information from these publications has been encoded into a Microsoft ACCESS® database. Database fields include species distribution by EPA Region, State, and specific locality for which the model was produced; parameters required to compute the HSI; and prioritized methods that can be used to obtain data to parameterize the models. Equations to calculate relationships of parameters (e.g., percentage canopy cover) to variables and the algorithms that combine variables into HSI values have been encoded into MS Excel® spreadsheets. Built in queries permit searches on any species or list of species to generate a compiled report of all potential species in a project area and the level of overlapping information

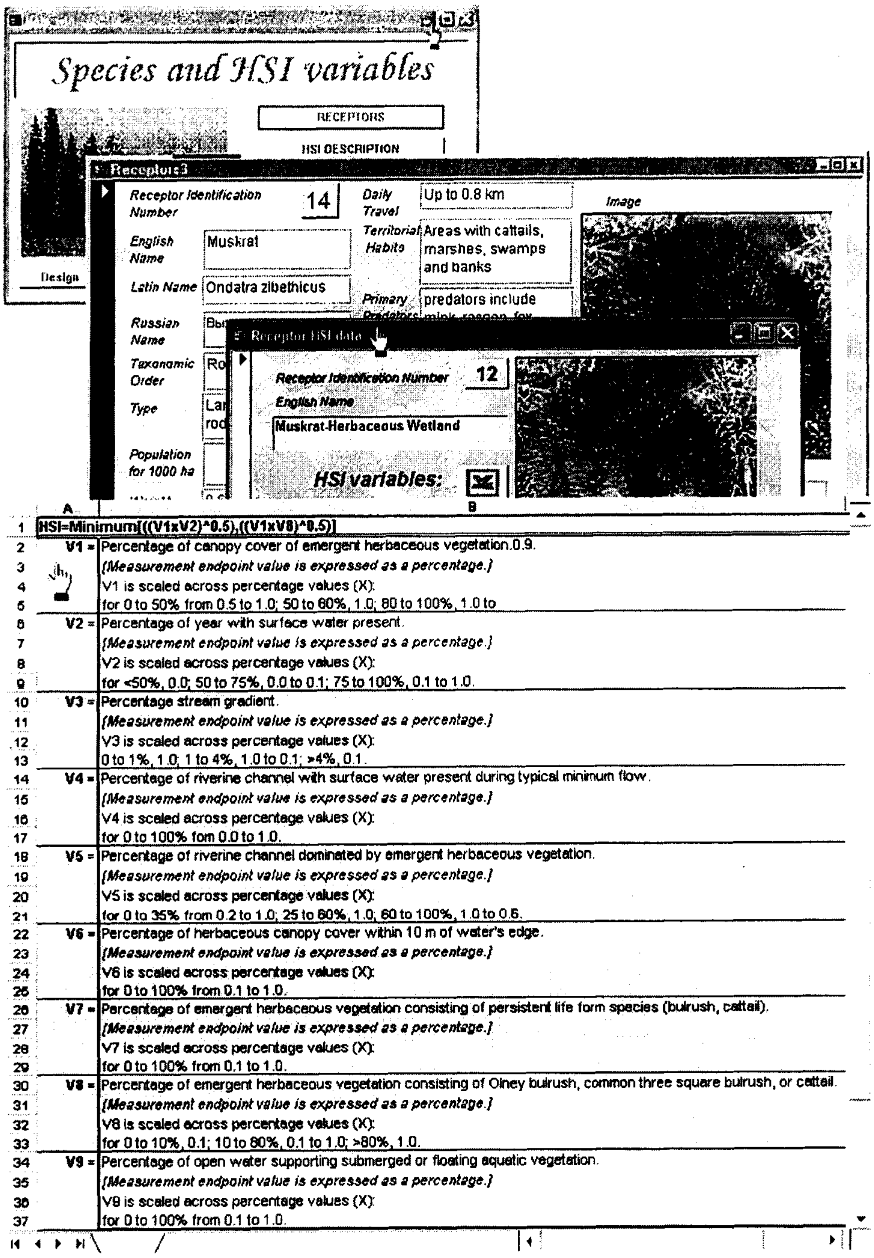


Figure 1. Example of HSI Database Windows

for each taxon in terms of habitat and dietary preferences

We plan to add many additional models. In total, published HSI models exist for 169 primary and overlap bird species. For the bird HSI model species, we have identified an additional 107 "overlap species." These are species for which no HSI models exist but for which, because of close similarities in their habitat requirements, existing individual HSI models may be appropriate (perhaps after modification).

4. Risk Trace Software

Risk Trace software described in Linkov et al., 2004 utilizes the approach used to design the spatial foraging sub-model in Linkov et al. 2002. The analysis employs a spatially explicit foraging sub-model that provides a time series of contaminant concentrations in soil and forage that a receptor may encounter within its habitat. The habitat is divided into a grid of one-meter by one-meter cells. Contaminant concentrations are then assigned to each cell based on site-specific measurements or GIS coverage. The spatial sub-model uses the habitat grid to calculate exposure point concentrations for a receptor via soil and plant pathways. The probabilistic receptor migration sub-model then generates random receptor movements to model which exposure concentrations the receptors will encounter. In general, receptors are modeled to prefer areas with high habitat quality; i.e., they move in preferred directions that are determined by location, attractiveness of habitat and forage resources. The rate of receptor migration within a habitat is inversely proportional to the forage volume and habitat quality of the surrounding cells. A probability of random movements is also assigned: at specified time periods, each individual receptor in the simulation is modeled foraging in randomly selected areas within the habitat.

Prior Risk Trace applications use habitat quality values that were based on expert judgment elicitation. The relationship between receptor migration and habitat quality was used as calibrating factor in model simulations. The approach that we present in this paper replaced potentially biased assignment of habitat quality by experts by a more rigorous and regulatory accepted habitat suitability modeling. When used with Risk Trace, the habitat quality index value becomes the determining factor for the movement and foraging behavior of the species for estimating exposure.

5. HSI Model Implementation within Risk Trace

The HSI sub-module within Risk Trace was developed within Microsoft Office and functions as a Microsoft Excel macro (a subprogram). It uses Visual Basic and FORTRAN to perform calculations and data processing. The user interface was developed using Visual Basic and is compatible with Microsoft Office. Through this interface, a user can develop scenarios and specify model parameters. The visual interfaces developed for the prototype version include:

- General site information.
- Receptor selection window and link to receptor database that offers relevant information as specified in paragraph 3.

- Site delineation window that allows specification and edition of habitat location and area of migration.
- HSI parameters insertion window that is to input the variables of HSI model for the species selected and habitat specified.
- Calculation of HSI values and visualization of results in a form of report.

The interface window shown in Figure 2 allows the user to select species and to delineate polygons, i.e., specify the spatial characteristics and locations of the identified migration zones overlying a map of the entire habitat. The user can change the location, shape, and extent of these areas using visual "drag-&-drop" procedure.

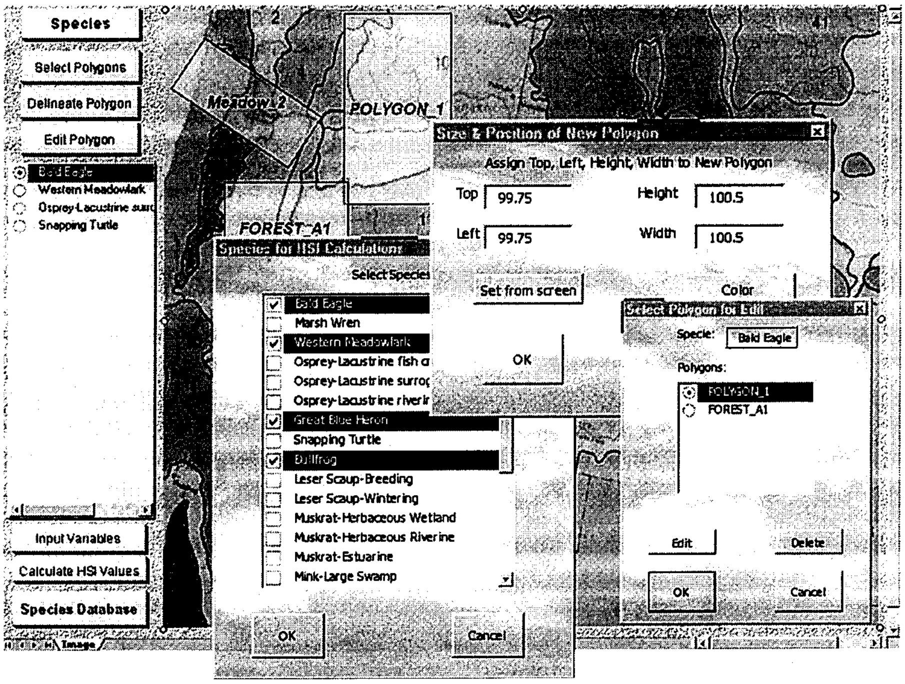


Figure 2. Main Menu with Species Selection and Polygon Delineation Windows

The interface window allows the user to specify site specific information for the parameters needed to calculate habitat quality for each delineated polygons for each receptor. For example, HSI calculations for Bald Eagle (Figure 3) require information on: (i) area covered by open water and adjacent wetlands; (ii) The Morphoedaphic Index, (iii) percentage of potential nesting area covered by mature timber; and (iv) density of building and campsites in the area.

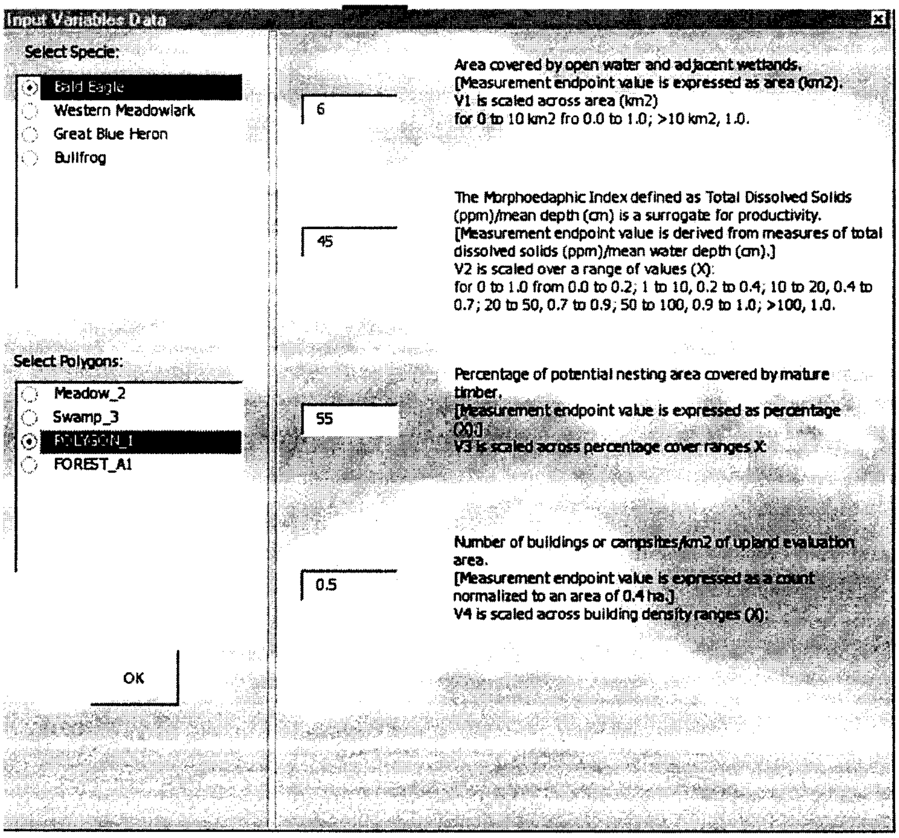


Figure 3. HSI Input Data for Bald Eagle

Based on this information, Risk Trace generates report were HSI values are calculated for each delineated polygon. Report also presents HSI model used in calculation as well as input parameters (Figure 4).

6. Application Example

At present, only an hypothetical example is developed to provide an illustration of how habitat quality would alter the exposure estimates. Prior to considering the HSI models as a means of describing habitat quality, a test of performance of the Risk-Trace portion of the model was performed using data for roe deer migration in an artificial landscape. Several landscapes with the same level of contamination and total area of high habitat were generated. In one extreme case, the areas of attractive habitat consisted of three squares (Fig. 5a), in the other extreme, three thin strips were modeled (Fig. 5b). Four additional landscapes with varying width/length ratios were also studied. Figure 5

displays a sample graphic output showing contamination and habitat maps for the two extreme cases of the modeled artificial landscapes. The contaminated zones are shown as solid lines. These zones are also assumed to be of a high habitat quality. Each dot in the figure presents the location of one of the 20 modeled receptors at a different time. In both the square and rectangular habitat presented, receptors migrate extensively within the zones with high habitat quality. For the time of simulation (180 days) receptor forage extensively in the northern portion of all three zones and in the middle zone (Fig. 5a), while more extensive foraging in the eastern zone was modeled for square patches (Fig. 5b).

Project Identification: HSI Demonstration

Site Name: Ochsenhausen 97 27 02 2002

Species: Bald Eagle

HSI=((V1xV2)^0.5)x((V3xV4)^0.5)

V1 = Area covered by open water and adjacent wetlands.
[Measurement endpoint value is expressed as area (km2).
V1 is scaled across area (km2)
for 0 to 10 km2 fro 0.0 to 1.0; >10 km2, 1.0.

V2 = The Morphoedaphic Index defined as Total Dissolved Solids (ppm)/mean depth (cm) is a surrogate for productivity.
[Measurement endpoint value is derived from measures of total dissolved solids (ppm)/mean water depth (cm).]
V2 is scaled over a range of values (X):
for 0 to 1.0 from 0.0 to 0.2; 1 to 10, 0.2 to 0.4; 10 to 20, 0.4 to 0.7; 20 to 50, 0.7 to 0.8; 50 to 100, 0.9 to 1.0; >100, 1.0.

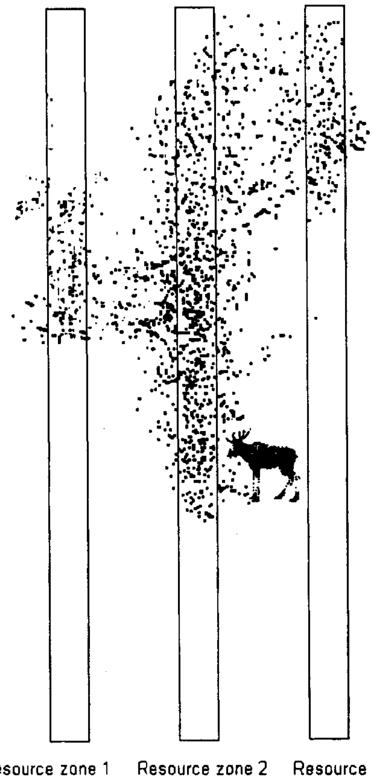
V3 = Percentage of potential nesting area covered by mature timber.
[Measurement endpoint value is expressed as percentage (X).]
V3 is scaled across percentage cover ranges X:
for 0 to 75% from 0 to 1.0; >75%, 1.0.

V4 = Number of buildings or campsites/km2 of upland evaluation area
[Measurement endpoint value is expressed as a count normalized to an area of 0.4 ha.]
V4 is scaled across building density ranges (X):
for 0 to 20 buildings/km2 from 1.0 to 0.0; >20, 0.0.

Polygon Identification	V1 Measurement Endpoint	V2 Measurement Endpoint	V3 Measurement Endpoint	V4 Measurement Endpoint	HSI
FOREST_A1	9,0	65	65,0	1,2	
	0,90	0,93	0,86	0,94	0,82
POLYGON_1	6,0	45	55,0	0,5	
	0,60	0,87	0,73	0,98	0,61
Swamp_3	1,0	56	10,0	0,8	
	0,10	0,91	0,13	0,96	0,11
Meadow_2	1,0	28	99,0	0,5	
	0,10	0,75	1,00	0,98	0,27

Figure 4. HSI Report for Bald Eagle

Radionuclide accumulation resulting from receptor migration in an artificial landscape with patches of different shape is high in the landscape with square patches and is lower in the landscape with more fragmented, thinner patches (Figure 6). In the landscape with square patches, the receptor migrates continuously in the area of high contamination and accumulates significant amount of radioactivity. In fragmented habitats, the likelihood of migration in the contaminated field is smaller. Because more animals could be exposed to different contamination levels (distributed nature of the contaminants) the overall uncertainty in the mean accumulation level (expressed as confidence interval) is higher for the patchy landscape



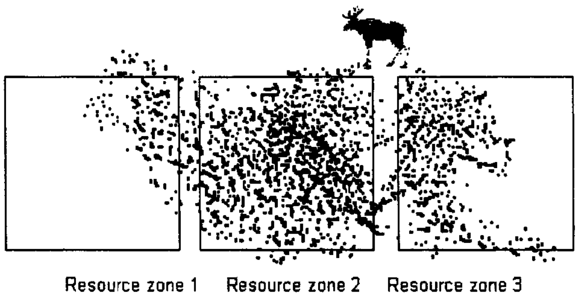


Figure 5. Receptor Migration Pattern in Artificial Landscape

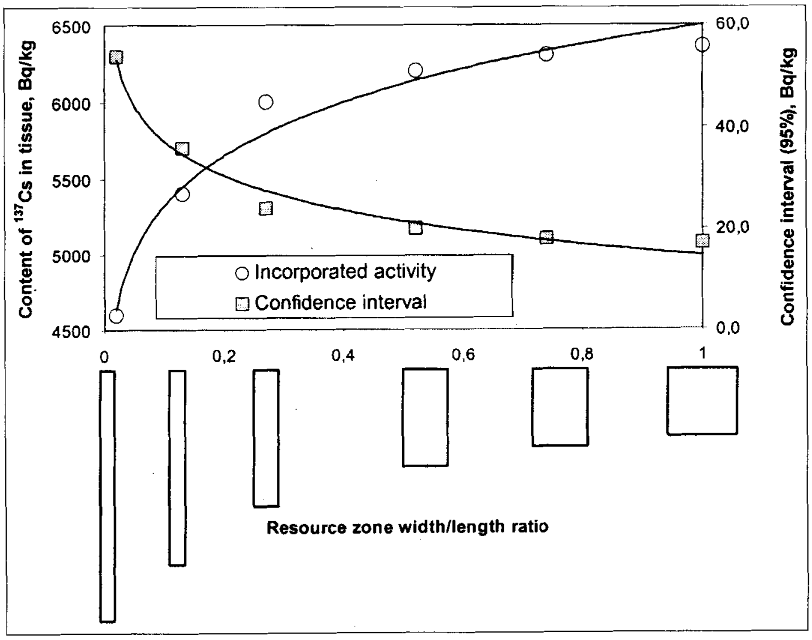


Figure 6. Cs Accumulation by Roe Deer Foraging in Landscapes with Different Patch Shapes.

7. Conclusion

The HSI models provide a controlled means of accounting for habitat conditions without requiring excessive costs. The qualitative differences that occur in landscape features under various remediation scenarios provide broad characterization of the effects in terms of habitat quality for the wildlife species of interest. When viewed in relative terms, rather than absolute quantitation of species abundance, the HSI model output can be a valuable tool in EcoRA (Kapustka et al., 2001, in press-a, in press-b, Kapustka, 2003).

We developed a spatially explicit foraging model that provides a time series of media and forage contamination that receptors may encounter during their daily movements. The model currently incorporates information on: geospatial parameters of the contaminated area, surrounding land, and habitat types found in each; density and distribution of ecological receptors; receptor home range; maps of contamination concentrations and habitat disturbance; and size of the receptor's foraging range. The model also employs habitat quality factors that account for differential attraction to various habitat types within the site. The model is developed for both terrestrial and aquatic ecosystems.

This paper is a part of our overall effort to incorporate spatially explicit ecological risk assessment into a risk-based protocol to be used in decision-making regarding the reuse or sustainable use of disturbed sites. We propose to approach these complex problems by combining the approaches from traditionally disparate schools of assessment. The tools and methodologies to be developed will incorporate concepts from both risk assessment and ecological assessment to simultaneously address the factors (e.g., pollutants) that decision makers need to eliminate or minimize and the factors (e.g., habitat, rare species) decision makers want to maximize. Further development of the risk-based protocols and related prototype software will:

- Further develop risk assessment algorithms;
- Make direct use of geographic information systems (GIS) technology, and further integrate data with GIS;
- Supplement the database with profiles for a wider range of receptors;
- Enhance the current default database of exposure parameters and risk benchmarks;
- Expand functional modeling capabilities to include food chains and other dynamic factors of the specific ecological situation; and
- Link the user to expert decision support systems.

8. Acknowledgement

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USE OF GIS AS A SUPPORTING TOOL FOR ENVIRONMENTAL RISK ASSESSMENT AND EMERGENCY RESPONSE PLANS

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Abstract

Although occurrence of disasters cannot be prevented completely, it is possible to minimize their hazards by taking precautions and applying effective emergency response plans. In addition to measures taken to reduce economical and human losses, an environmental dimension is required in these plans to control environmental pollution and lessen possible adverse effects on both ecosystems and human health, which in the long term may cost much more than direct disaster losses. Technological accidents triggered by natural disasters are one of the most important factors increasing the environmental damage. Therefore, it is of utmost importance to prepare regional plans considering both natural and technological disasters and aiming the coordination and resource sharing between the related authorities, institutions and factories. Geographical Information Systems (GIS) are powerful tools having comprehensive data query, analysis, and visualization capabilities, and they may facilitate preparation of such emergency plans. In this paper, the role of GIS in emergency response plans is explained. A case study from Turkey utilizing GIS extensively for regional environmental emergency planning is given and problems that can be faced in developing countries are discussed.

1. Introduction

Natural disasters that result in loss of many human lives and cause tremendous economical loses continue to be a major problem for mankind. In addition, large-scale environmental damages are also generally inevitable as consequences of disasters. Although most environmental damages are associated with the nature of the disaster,

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resulting secondary technological accidents may also cause a considerable increment in damage. Industries located within the disaster region and that deal with hazardous chemicals, like chemical production or hazardous waste storage facilities, are especially important due to their potential to increase the degree of environmental damage. In such cases, the additional adverse effects on human health and ecosystems may be heavier in long term than the direct losses of the disaster. Their additional effects, which may be a result of chemical spillage to the environment, fire, or explosion, depend mainly on the degree of hazard that the facility has been subjected to during the natural disaster, and the nature and amount of the chemical.

There is always an occurrence risk of disasters and accidents, and these cannot be prevented completely. However, it might be possible to minimize and control the environmental pollution by preparing for such events prior to such an occurrence, and by a rapid response afterwards. Although the standard required control measures for chemical accidents may be taken and facility-wise emergency response plans may be prepared, they may lose their effectiveness owing to the extraordinary conditions faced during a major disaster. For example, the resources required for emergency response within the facility might have to be used for local emergency response to the natural disaster. Or the people themselves and their families may be at vital risk, which complicates the decisions regarding priorities. Similar problems are also likely for natural disaster emergency response plans that do not take secondary technological accidents into account. Although the effects of the technological accidents seem to be lesser at the first sight, the overall effect may be much more under certain conditions and should not be underestimated.

For these reasons, preparation of effective emergency response plans that consider both natural disasters and technological accidents and that aim at coordination between the related authorities, institutions, and facilities is extremely important. Although examples of such plans exist in developed countries, they are still lacking in developing countries.

2. Emergency Response Plans and Geographical Information Systems

Preparation of such emergency response plans has several steps, which can be summarized as in Figure 1. In each phase there exists a high amount of uncertainty, which complicates the decision making process. Having as much and as recent information as possible at hand facilitates the decisions. However, high amounts of information may become unmanageable, especially if the disaster region is broad in extent.

Computer-aided systems like databases and decision support systems can be very helpful in such situations. One of these technologies, Geographic Information Systems (GIS), with its ability to relate geographical and attribute data, can be an effective and efficient platform for the management of information. GIS provides a means of rapid data access and query based on both geographic location and attribute data. Using GIS mapping functions, it is possible to superimpose two or more data layers and to relate otherwise disparate data on the basis of common geographic location. Because GIS products can be produced quickly, multiple scenarios can be

evaluated in a short time. The advanced data visualization capabilities of GIS may also facilitate communication in the case of an emergency. Data in the form of a map is much easier to interpret than data in a table. Similarly, outputs from risk assessment studies can be more clearly presented to decision makers using GIS maps.

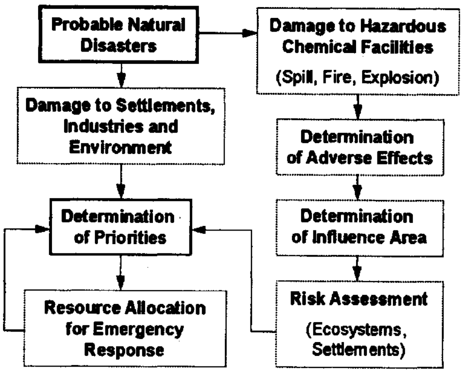

Fig 1. Emergency response phases

A GIS developed to support environmental emergency response plans should be capable of;

- Forming relations and data structures for effective collection and storage of the information that is required for development of the emergency plan and risk assessment phases;
- Representing and visualizing the information in an user-friendly environment so that data access is facilitated;
- Providing means for efficient data update;
- Easily conducting required spatial analyses like overlay analysis, network analysis and buffering; and
- Supporting integration with other supporting systems and modeling tools.

3. A Case Study

Turkey is a country that has endured significant loss of life and property due to natural disasters. Due to the recent earthquakes in 1999 alone, more than 18,000 people died. The economical losses caused by these earthquakes are estimated to be 10-15 billion USD [1]. In Turkey, the first comprehensive legislative approach to disaster management dates back to 1959. This legislation, which is still is use, was designed to address all kind of natural disasters, but it focuses mainly on earthquakes since they are the most destructive natural events that Turkey experiences. One of the major weaknesses of this legislation is that it does not cover technological disasters. (There is also no other legislation relating to the management of this kind of disaster.) Another

weakness is that it does not take the environmental aspects of disasters into consideration. The legislation does not describe any precautions or arrangements to deal with environmental problems that may occur as a result of natural disasters.

The Marmara Region, being the most densely industrialized region of Turkey and carrying a first-degree risk of earthquakes, requires the highest priority in the preparation of emergency response plans considering environmental issues. This region, in which one third of national industrial production occurs, also has the highest population density in the country. More than 10 million people live in Istanbul alone.

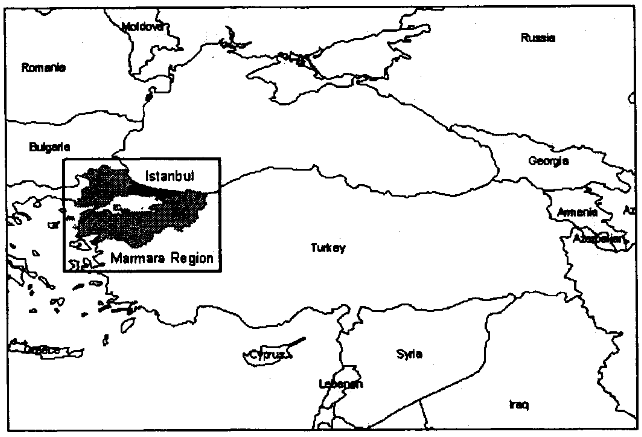

Fig 2. The Marmara region

Recent earthquakes in the region, in combination with environmental damages that occurred afterwards, have indicated that there is an urgent need for the development of a regional emergency response plan that covers environmental impacts. Furthermore, it must be recognized that these impacts are not only the direct ones but also the impacts of industrial accidents or disasters caused by natural disasters. In the year 2000, the Turkish Ministry of Environment initiated the preparation of an Environmental Emergency Response Plan (EERP) for the Marmara Region and requested that our group develop a GIS-based database to be used in conjunction with the EERP.

The database we developed includes information on environmental resources, meteorology, industrial facilities, and the hazardous chemicals that are produced, consumed, stored or wasted by these facilities. The capabilities of the system include:

- Grouping of the facilities according to their chemicals, wastes, and degree of natural hazard risks by a city and/or regional basis;
- Determination of industries that pose risk to a sensitive environment in case of an accident;
- Determination of sensitive ecosystems that may be exposed to risk due to a specified facility in case of an accident;

- Access to the inventory of the hazardous chemicals in a facility to determine specific risks that may occur as a result of a disaster;
- Access to safety information on hazardous chemicals in the format of material safety data sheets (MSDSs);
- Access to individual emergency response plans for the included facilities;
- Provision of historical meteorological data for assessment of the atmospheric dispersion of chemicals.

In order to develop the database, a base map was first prepared: this included administrative boundaries, settlements, transportation networks, and establishments dealing with hazardous chemicals. Next, we created the structure for data storage on industrial establishments. This data structure includes general information about the establishment (such as name, address, telephone and fax numbers, etc.), chemicals that are produced, utilized or stored, and wastes that are generated by the establishment. The emergency response plans of the establishments and a list of persons responsible in an emergency are also included.

The hazardous chemicals section, which is divided into four main categories (produced, consumed, stored and wasted), includes for each chemical its common name, amount, phase, concentration, CAS number, and code number according to the Turkish Regulation of Hazardous Chemicals. Additional information, such as storage conditions, storage capacity, and type of waste management, are given for specific categories. In order to inform the user of specific health risks and suggest measures that should be taken to reduce these risks, MSDSs for chemicals are integrated in the database. Physicochemical properties of the chemicals are also supplied to aid environmental fate modeling.

The database also includes information on natural hazards and natural resources in the region that can be affected by these hazards. Listed in the order of most destructive to least, earthquakes, floods, forest fires, landslides, and rock falls are natural disasters that are important for Turkey and that occur frequently. Maps integrated into the system include epicenters and magnitudes of earthquakes that have occurred in the last hundred years [2], active earthquake faults [3], probabilistic earthquake risk zones [4], forest fire sensitivity zones [5], and flood zones for last thirty years [6]. The environmental resources map mainly includes information on surface and groundwater resources. In particular, the surface water resources map consists of watershed boundaries, rivers, lakes, dams, and monitoring stations located in these water bodies. The boundaries of groundwater aquifers and locations of ground water wells are also given as sub-surface information. Monthly averages of meteorological data are supplied with the database to aid the assessment of atmospheric dispersion of chemicals (the most important transport mechanism for volatile hazardous chemicals). A digital elevation model of the region is also supplied.

To facilitate data access, we constructed an electronic facility information form. Using this, the user can enter and access comprehensive information on hazardous chemicals that the facility deals with and obtain maps of settlements, natural resources and hazard risks in the vicinity of the establishment. The details of the establishment information form are given in Figure 3.

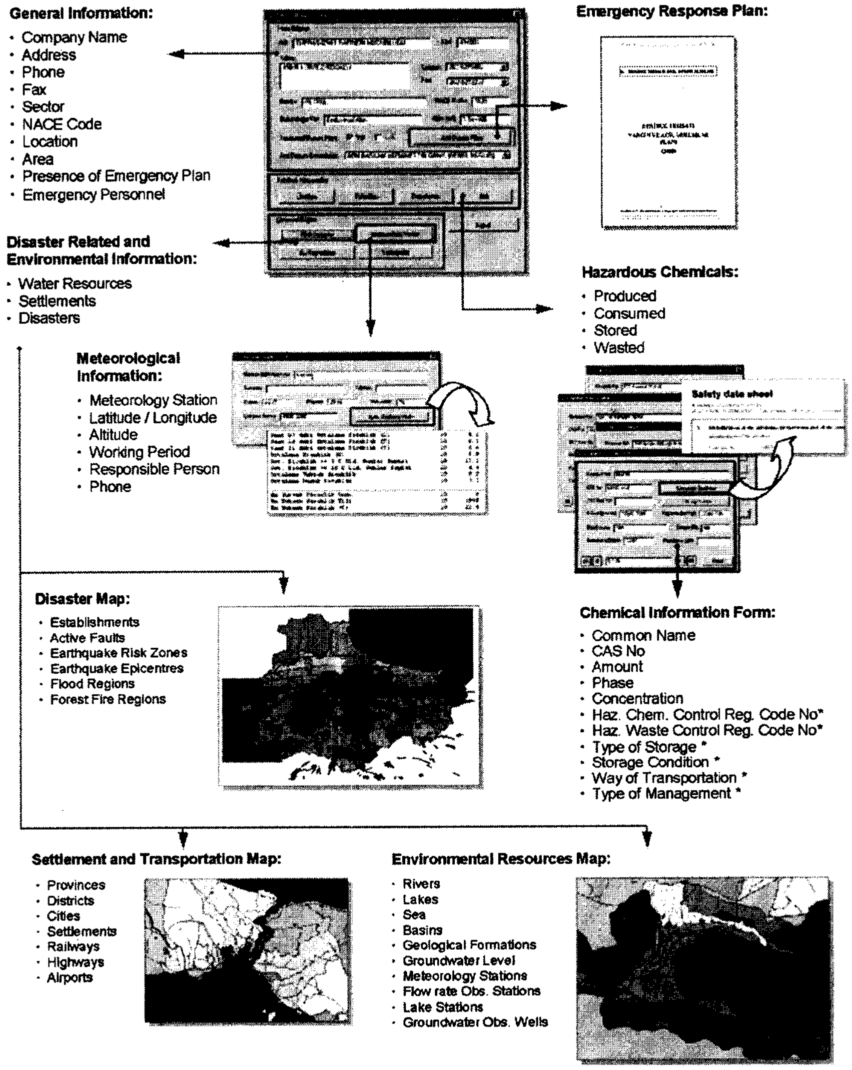

Fig 3. Establishment Information Form

3. Discussions

There are several problems that could be faced during the development of a large-scale geographical database like the one given in the case study. The first problem is that of

data availability. Unfortunately, in most developing countries, many kinds of environmental data are not available. In particular, digital data is rarely found. This is mainly due to the limited economic resources that can be allocated to systematic measurements and analyses, as well as the lack of continuity of any such measurements started previously.

The second major problem is that of data quality and standardization. Data, if it exists, is generally far from being in a standardized format. This limits its usage, especially for time- or location-based comparisons. In addition, data quality measures are mostly unavailable, leading to possible data unreliability.

Another important problem is obtaining up-to-date data. In order to assess risk in a reliable manner, the data used for assessment should reflect actual conditions as much as possible. This is especially important for emergency cases. During a technological disaster, an evaluation of possible adverse effects of a hazardous chemical that uses as an input the amount that an establishment had one year ago may lead to large deviations from reality and dramatic results at the conclusion. Therefore, data should be updated regularly by the individual facilities. But, as stated in the discussion of data availability above, this cannot be guaranteed for developing countries most of the time.

Several extensions to our developed geographical database are possible. Currently the system can be used only as an information source with advanced query and analysis capabilities. By integrating environmental fate models into the system, the distribution of hazardous chemicals in the environment, especially in the atmosphere and in water resources, could be predicted. This information could then be used to determine the vulnerability of specific settlements and environmental resources due to technological accidents. Or, in order to solve the problem of keeping data updated, a real-time connection to data sources could be provided. Although not possible for all of the incorporated data types, such a connection might be especially possible for meteorological measurements. Finally, an Internet interface through which the facilities could update their hazardous chemical inventory could be very helpful.

4. Conclusions

Although occurrence of disasters cannot be prevented, it is possible to minimize their hazards if precautions are taken and emergency response plans are applied. An environmental dimension is required in the emergency response plans to minimize and control environmental damage and pollution and their adverse effects on human health and ecosystems. GIS, provided that enough data exists for such a system, is a powerful tool for risk assessment and decision-making in such cases, due to its comprehensive data query, analysis, and visualization capabilities. In the present study, this technology has been extensively employed and a regional information system for the Marmara area of Turkey, including Istanbul, has been developed. Use of GIS makes it possible to easily determine the residential areas and environmental resources around facilities that may be affected by a disaster. The methodology used for the development of this system is applicable directly to other regions and probably to other countries.

The solution to the problems faced during development of such systems starts with standardization of data formats and development of national databases. Once such resources are created, this type of database can provide fast and easy data access to decision-makers, scientists, and the public. With cooperation between the institutions, such databases may be linked to each other, and could form a backbone for a national information system. As the appreciation of the environment increases, problems of data availability and quality can be solved more easily. This will require some time for developing countries, since for these regions the facility productivity and the economic growth currently have a higher priority than the environment.

5. Acknowledgements

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INTEGRATED RISK ANALYSIS FOR SUSTAINABLE WATER RESOURCES MANAGEMENT

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Abstract

Two main criteria are usually taken into consideration for engineering water resources management, namely technical reliability and economic efficiency (techno-economic approach). To obtain sustainability, one should consider not only technical and economic issues, but also environmental and social aspects. In this paper it is explained how integrated risk analysis considering risk indexes in four dimensions (technical, economical, environmental and social) can be used in order to quantify the degree of sustainability in water resources management.

1. Introduction

Water resources management involves different disciplines, such as engineering, chemistry, ecology, economy, law and social sciences. Traditionally, the general objective of water management has been the satisfaction of demand for various uses, such as agriculture, drinking water or industry, using available water resources in technically reliable and economically efficient ways. This approach has led to structural and mostly technocratic solutions being suggested and implemented in several countries. However, in many cases building dams modifying riverbeds and diverting rivers has had serious negative repercussions on the environment and on social conditions. Moreover, waste in the use of this precious resource and rampant pollution in all areas of water use, have raised doubts about this form of management. The concept of a sustainable management of water resources was first mentioned in Stockholm in 1972, during the United Nations World Conference and then at the Rio summit in 1992 with Agenda 21.

The new philosophy is based on the integrated management of water at the watershed basin level. Emphasis is placed on environmental protection, the active participation of local communities, demand management, institutional aspects and the role of continuous and lifelong education of all water users.

On the methodological level, integrated water management remains an open question and several different approaches seek to define a coherent paradigm. One possible paradigm is proposed in this paper and may be called the «4E paradigm»: Epistemic, Economic, Environmental, and Equitable. It is based on integrated risk

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analysis, with a multidimensional characterization of different risks: scientific, economic, environmental and social. This paradigm uses either the theory of probability, or fuzzy logic, or both in order to assess and integrate technico-economic and socio-environmental risks in a perspective of sustainable management of water resources.

The aim of this paper is to show how traditional engineering planning and design methods for reducing risks in water supply and management can be extended to consider environmental and social risks. Furthermore, a multiobjective decision-making methodology is suggested, in order to rank different feasible solutions.

2. Risk Definition and Methodologies for Risk Quantification

Engineering risk and reliability analysis provides a general framework to identify uncertainties and quantify risks. As shown in this paper, so far two main methodologies have been developed to assess risks (Ganoulis, 1994):

(a) the stochastic approach, and
(b) the fuzzy set theory.

Stochastic variables and probability concepts are based on frequency analysis and require large amounts of data. Questions of independence between random variables and validation of stochastic relations, such as the well-known statistical regression, are often difficult to resolve. Fuzzy set theory and fuzzy calculus may be used as a background to what could be called "imprecision risk analysis". In this paper it is demonstrated how fuzzy numbers and variables may be used for estimating risks in cases where there is a lack of information or very little data available.

2.1 THE TECHNOCRATIC APPROACH

Depending on the particular sector involved (e.g. drinking water, hydraulic engineering, agriculture or industry), various engineering specialties have been developed to address problems of water resources management.

From a traditional and rather 'technocratic' point of view, water resources planning may be defined as the process of developing alternative water quantities in order to satisfy demand over a given time period.

This technocratic concept reduces water supply questions to the mere technical problem of collecting and distributing water volumes, in order to satisfy different water demands for drinking, irrigation or industrial use. Engineering planning and the design of structural or non-structural alternative solutions are the usual tools used by this profession for water management studies.

The different steps involved in engineering planning are indicated in Fig. 1. The first step is to provide alternative technical solutions, by the use of data and mathematical modelling. Then, the decision making process is developed by introducing different criteria, such as technical reliability and economic efficiency.

2.2 COGNITIVE AND NON-COGNITIVE UNCERTAINTIES

Uncertainties are actually due to a lack of knowledge about the structure of various physical and biochemical processes, and to the limited amount of data available ([2], [16], [5], [7]). Several authors have analysed different types of uncertainties and distinguished between uncertainties, which may be objective or subjective, basic or secondary and natural or technological.

Another distinction should also be made between (1) ***non-cognitive*** or natural uncertainties or randomness, and (2) ***cognitive*** or man-induced or technological uncertainties.

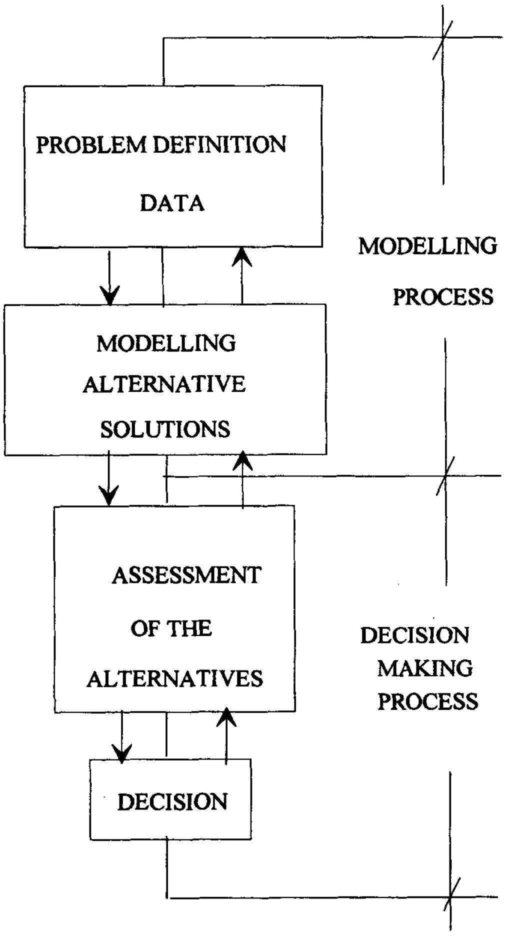

Fig. 1: Steps in engineering planning.

2.2.1 Non-cognitive uncertainties or randomness
It is postulated that natural uncertainties are inherent to a specific process, and that they cannot be reduced by using an improved method or more sophisticated models. Uncertainties due to natural randomness or non-cognitive uncertainties may be taken into account by using stochastic or fuzzy logic-based methodologies, which are able to quantify uncertainties.

2.2.2 Cognitive or man-induced uncertainties
Man-induced uncertainties are of different kinds: (a) data uncertainties, due to sampling methods (statistical characteristics), measurement errors and methods of data analysis, (b) modelling uncertainties, due to the inadequacy of the mathematical models in use and to errors in parameter estimation, and (c) operational uncertainties, which are related generally to the construction, maintenance and operation of engineering works. Contrary to natural randomness, cognitive uncertainties may be reduced by collecting more information, or by improving the mathematical model being used.

2.3 DETERMINISTIC, STOCHASTIC AND FUZZY VARIABLES

Although rather exceptional, there are situations in water resources engineering which can be considered as deterministic. In such cases mathematical deterministic approaches relating inputs to outputs are sufficient, because uncertainties are low. Take, for example, the effect on water flow rate from a reservoir by changing the reservoir water level. There is a deterministic relation between the flow rate and the water level in the reservoir. In such a case risk and reliability techniques should not be used, because the situation is predictable.

When the reservoir is filled by an inflow that varies randomly in time, various uncertainties produce a variation of water level in the reservoir that is no longer deterministic. It may be considered as a stochastic or probabilistic variable.

Imprecision in boundary conditions and modelling coefficients can be quantified and propagated by use of fuzzy numbers and fuzzy logic-based modelling [8].

2.4 DEFINITION OF THE ENGINEERING RISK

In a typical problem of technical failure under conditions of uncertainty, there are three main questions, which may be addressed in three successive steps.

1. When should the system fail?
2. How often is failure expected?
3. What are the likely consequences?

The first two steps are part of the uncertainty analysis of the system. The answer to question 1 is given by the formulation of a critical condition, producing the failure of the system. To find an adequate answer to question 2 it is necessary to consider the frequency or the likelihood of failure. This can be done by use of the probability calculus. Consequences from failure (question 3) may be accounted in terms of economic losses or profits.

It has been largely accepted that the simple definition of the engineering risk as ***the probability of failure*** (risk = probability) is much more appropriate.

As shown in Fig. 2, the actual approach in engineering water resources planning methodology aims primarily to reduce technical and economic risks by achieving two main objectives

- Technical reliability or performance, and
- Economic effectiveness.

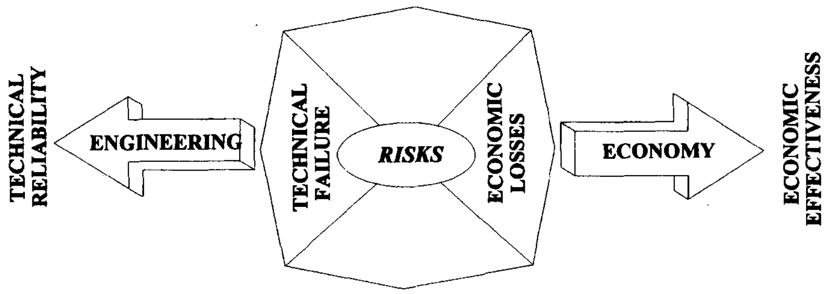

Fig. 2: Technical and economic performances in water resources management.

As explained in [7] we should define as *load* λ a variable reflecting the behaviour of the system under certain external conditions of stress or loading. There is a characteristic variable describing the capacity of the system to overcome this external load. We should call this system variable *resistance* r. A *failure* or an *incident* occurs when the load exceeds this resistance, i.e.,

FAILURE or INCIDENT : $\lambda > r$

SAFETY or RELIABILITY : $\lambda \leq r$

In a probabilistic framework, λ and r are taken as random or stochastic variables. In probabilistic terms, the chance of failure occurring is generally defined as risk. In this case we have

$$\text{RISK} = \text{probability of failure} = P(\lambda > r)$$

2.5.INSTITUTIONAL AND SOCIAL ISSUES

In recent years special attention has been paid to institutional and social approaches in water resources management and flood alleviation planning [8]. The institutional or administrative framework may be conceived as being the set of state owned agencies or private enterprises dealing with production, distribution and treatment of water.

Of particular importance is their scale of operation (local, regional or state), their degree of autonomy from the central administrative body, and the involvement of different water stakeholders in the decision making process. The administrative systems and water laws and regulations, together with social perception on the use of water and traditions involved, make the issue of water resources management very complex.

2.6 FUZZY LOGIC – BASED APPROACH

Consider now that the system has a resistance R and a load L, both represented by fuzzy numbers. A *reliability measure* or a *safety margin* of the system may be defined as being the difference between load and resistance ([7], [19]). This is also a fuzzy number given by

$$\widetilde{M} = \widetilde{R} - \widetilde{L}$$

Taking the h-level intervals of R and L as

$$R(h)=[R_1(h), R_2(h)], \qquad L(h)=[L_1(h), L_2(h)],$$

then, for every $h \in [0, 1]$, the *safety margin* M(h) is obtained by subtracting L(h) from R(h), i.e.

$$M(h) = R(h) - L(h).$$

Two limiting cases may be distinguished, as shown in Fig. 3:
There is *absolute safety* if:

$$M(h) \geq 0 \qquad \forall\, h\, [0,1]$$

whereas *absolute failure* occurs when:

$$M(h) < 0 \qquad \forall\, h\, [0,1]$$

A *fuzzy measure of risk*, or *fuzzy risk index* R_i may be defined as the area of the fuzzy safety margin, *where values of M are negative.* Mathematically, this may be shown as:

$$R_i = \frac{\int_{m<0} \mu_{\tilde{M}}(m)dm}{\int_{m} \mu_{\tilde{M}}(m)dm} \qquad (1)$$

The *fuzzy* measure of reliability, or fuzzy reliability index R_e is the complement of (1), i.e.

$$R_e = 1 - R_i = \frac{\int_{m>0} \mu_{\tilde{M}}(m)dm}{\int_{m} \mu_{\tilde{M}}(m)dm} \qquad (2)$$

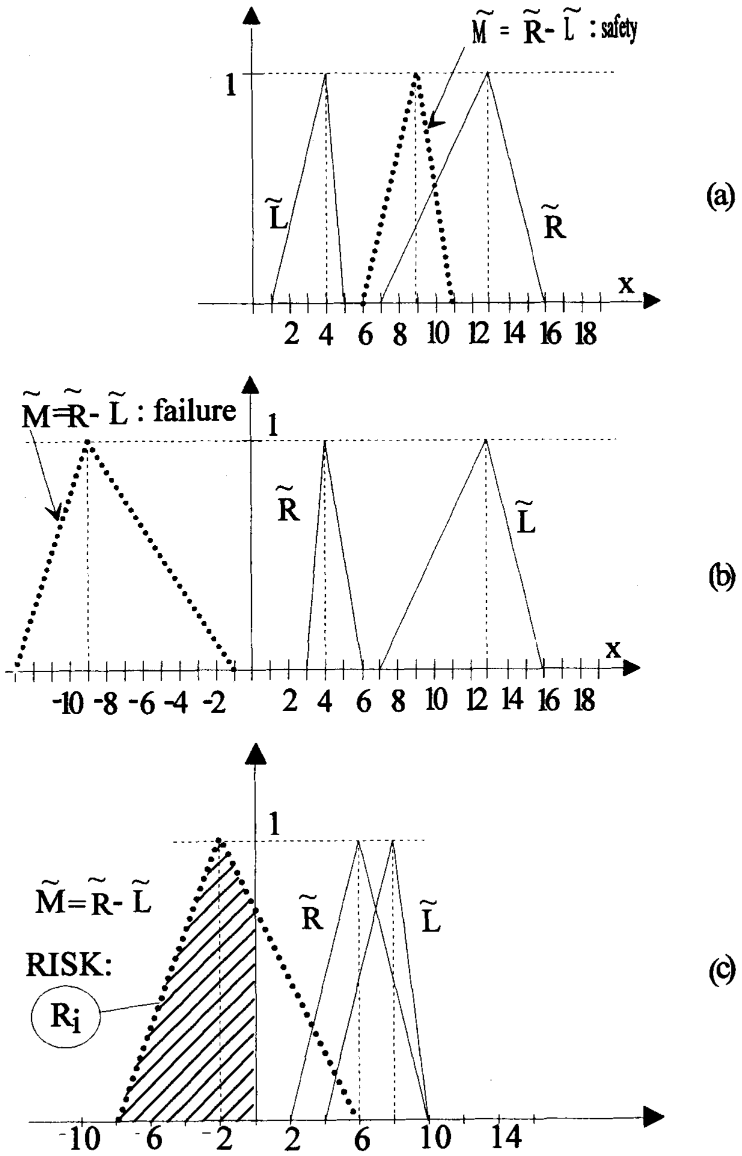

Fig. 3: **Absolute safety (a), absolute failure (b) and fuzzy risk (c).**

3. Fuzzy Modelling

Fuzzy modelling has not yet been developed extensively, although fuzzy numbers and fuzzy relations have found many applications in control engineering and industrial devices. Fuzzy set theory ([22], [13], [23]), and its derivative fuzzy arithmetic [12], may be used in order to introduce imprecise data into a mathematical model in a direct way with minimal input data requirements. In fuzzy modelling only the range and the most confident values of the input variables are required, so it can be used successfully when the available data is too sparse for a probabilistic method to be applied ([7] [20]).

In this model the various parameters and loads from external sources are considered as triangular fuzzy numbers (T.F.N.). In order to calculate the concentration of all pollutants at each node using finite differences or finite elements, a system of fuzzy equations needs to be solved. This is difficult from a mathematical point of view, and has stimulated a lot of interest because whatever possible technique is used, only enclosures for the range of the output variables can be produced.

Shafike ([18]) introduced the fuzzy set theory coupled with the finite element method into a groundwater flow model. The algebraic system of equations with fuzzy coefficients was solved with an iterative algorithm [14]. The fuzzy set theory was also applied into a steady-state groundwater flow model with fuzzy parameters combined with the finite difference method. A non-linear optimisation algorithm was used for the solution of the groundwater flow equations with fuzzy numbers as coefficients for the hydraulic heads.

Ganoulis et al. [9] used fuzzy arithmetic to simulate imprecise relations in ecological risk assessment and management. Specifically the technique was applied to a simplified domain with coastal circulation, in order to evaluate the risk of coastal pollution. For the solution of the algebraic system of equations with fuzzy coefficients direct interval operations were employed, instead of the iterative methods or non-linear optimisation techniques used in previous studies. Since triplets cannot be used for the multiplication and division operations, as explained in [12], mathematical operations have been performed at various h-level cuts by the use of the interval of confidence at each h-level.

It is also important to mention that the solution of an interval equation using interval operations is always an enclosure of the exact solution ([11], [14], [15]). The best possible enclosure for an interval function, which is defined as the "hull" of the solution, is a fundamental problem of Interval Analysis and should be treated with care, as the solution accuracy depends on the shape of the interval function [17].

The technique was tested initially with a one-dimensional advection-dispersion model in the non-conservative form, using the finite difference method. The results derived from the numerical computation considering the dispersion coefficient as fuzzy parameter are very similar to those of the analytical solution, confirming the accuracy of the numerical technique. A finite element algorithm combined with fuzzy analysis was also used for the solution of the advection-dispersion equation.

4. Towards a Sustainable Water Resources Management Approach

To integrate risk assessment into the socio-technical decision-making process of water resources management, the Multi-Risk Composite-Method (MRCM) is proposed. This is a variant of a Multi-Criteria Decision-Making methodology (MCDM). MCDM has been extensively used in the past for ranking different alternative options under multiple criteria or objectives.

Different analytical techniques for MCDM are available in literature ([10], [21], [4]). Recently, the following have received much more attention:

- ELECTRE I to III
- Compromise Programming
- Goal Programming
- Sequential Multiobjective Optimisation
- Game Theory.

In selecting the most appropriate method, important criteria are the kind of objectives (quantitative or qualitative), the number of decision-makers (one or a group) and whether objectives are involved a priori, a posteriori or interactively. ELECTRE I to III techniques are more suitable for qualitatively expressed criteria [1]. Game and team theories [3] are mainly interactive techniques.

Uncertainties and risk may be quantified by using probabilities or fuzzy sets, and can be handled better by Compromise Programming Techniques ([8], [6]). The Multi-Risk Composite-Method (MRCM) belongs to this kind of method.

As shown in Fig. 4, four main ***objectives*** or ***criteria*** are to be taken into consideration:

1. ***Engineering Reliability***: some measures for technical performance are: technical effectiveness, service performance, technical security, availability and resilience.
2. ***Environmental Safety***: environmental indicators may be positive or negative environmental impacts, such as increase or decrease in the number of species, public health issues, flora and fauna modifications, losses of wetlands, landscape modification.
3. ***Economic Effectiveness***: costs and benefits are accounted, such as project cost, operation and maintenance costs, external costs, reduction of damages benefits, land enhancement and other indirectbenefits.
4. ***Social Equity***: social impacts are, for example, related to risk of extremes, duration of construction, employment increase or decrease and impacts on transportation.

After the definition of the objectives, the steps to be undertaken for the Multi-Risk Composite Method planning are the following [7]:

1. Define a set of ***alternative actions*** or ***strategies***, which includes structural and non-structural engineering options.

2. Evaluate the outcome risks or ***risk matrix*** containing an estimation of the risks corresponding to each particular objective (technical, environmental, economic and social).

3. Find by use of an averaging algorithm the ***composite risk index*** for technical and ecological risks (eco-technical composite risk index) and the same for the social and economic risks (socio-economic composite risk index).

4. ***Rank the alternative actions***, using as criterion the distance of any option from the ideal point (zero risks).

As shown in Fig. 4, in the two-dimensional plane with coordinates the composite eco-technical and socio-economic indexes, strategies 1, 2 and 3 are ranked 1-3-2 using as criterion the distance of any strategy from the ideal point (0,0).

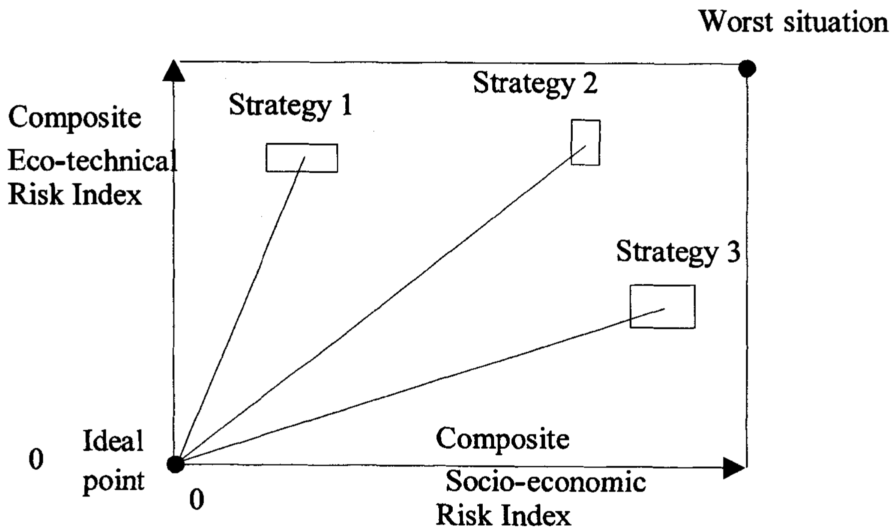

Fig. 4: Ranking different strategies based on eco-technical and socio-economic risks.

5. Conclusions

Integrating environmental and social issues into the engineering water resources management and flood alleviation planning is a challenge and a shift to a new scientific paradigm.

A methodology is proposed to integrate multiple risk analysis into a multi-objective planning and decision-making process. In order to assess and rank different alternative strategies for water resources management the methodology called Multi-Risk Composite Method takes into consideration four main objectives, namely technical, economic, environmental and social.

Ranking of different alternatives is based on the least distance from the ideal point of zero risk by use of two composite risk indexes:

1. the eco-technical risk, and
2. the socio-economic risk.

In order to achieve sustainability, by comparing this hydro-social approach to the traditional one of engineering water resources management, not only technical reliability and cost effectiveness are taken into account but also environmental safety and social equity.

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OVERCOMING UNCERTAINTIES IN RISK ANALYSIS: TRADE-OFFS AMONG METHODS OF UNCERTAINTY ANALYSIS

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Abstract

Industrial risk analysis suffers from many problems of uncertainty due to the difficulty of estimating various parameters of concern for the analysis. In the real world, we usually use many qualitative and/or uncertain parameters for risk evaluation. While the quantification of parameters is an important task, it is usually practiced according to the experience of the analyst (discrete approach) or by using probabilistic models (probabilistic approach). The discrete approach is very limited because it does not take into account the variability or the uncertainty of parameters. On the other hand, the probabilistic approach requires knowledge of the parameter's statistical distribution, which may be very difficult or even impossible. Furthermore, in both approaches, qualitative variables are not easy to deal with. Over years of research, we have developed a general approach to overcome these problems. The estimation of parameters and the treatment of available data are based upon fuzzy logic models, with some improvements in the fuzzy reasoning mechanism. This paper presents a comparison between our fuzzy approach and the discrete and probabilistic approaches. A geotechnical application was developed to evaluate the risk of natural ground movements in a rock cliff that would have severe impact on the surrounding environment. We have ended up with a general approach to the problem of uncertainty and with some recommendations on how to approach different parameters according to their nature (using either the discrete, probabilistic or fuzzy method). The improvements we have made to the fuzzy reasoning process (beta cuts reasoning technique) has been approved by specialists in the domain of fuzzy logic and are applicable to all branches of science.

1. Introduction

Since the end of the second World War, risk analysis studies have been carried out for many industrial systems (nuclear, chemical, etc.) and are usually more or less based on a two-dimensional space concept defined by the probability and the severity of an accident, as shown in Figure 1.

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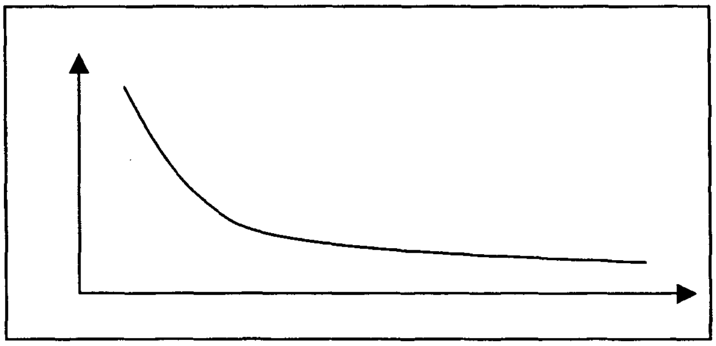

Fig. 1. Definition of risk as a two-dimensional function

Geotechnical engineering deals with nature and natural elements (rocks, soils, etc.). These natural elements are sometimes difficult to describe as quantified parameters. That is why it is necessary to construct a special methodology for geotechnical risk assessment that can take this consistent problem into account.

Care has to be taken not to mix up the study of slope or underground structure stabilities with risk analysis studies as done by Chowdhury et al. [1], Hudson et al. [2], Kawakami et al. [3], Nathanail et al. [4], Nguyen [5], and Chowdhury [6]. Most of these studies are concerned with the appropriateness of design parameters, stability and support, or the existence of potential hazards at a specific site.

Since our main concern is to establish a general risk analysis methodology for geotechnical work, we have decided to combine probabilistic factors with sensitive parameters in the hopes of implementing a probabilistic risk concept.

2. Uncertainty Analysis

At present, engineers mainly use the discrete approach when carrying out geotechnical risk analyses. This is a relatively old and simple approach. Whenever the uncertainty of a parameter's value is faced, an expert has to estimate a unique value, or point estimate, for the parameter. This estimate is usually based on the expert's experience, or it may be supported by few measurements. The estimation might be pessimistic (in the case of a minimum value), optimistic (a maximum value), or average (a mean value). The analysis of the problem is carried out by making point estimates for each parameter and using these values in the analysis,, which leads to a unique final result, in the form of a point estimate, for a specific problem.

This approach draws no attention to the nature of the parameter, nor to its probabilistic distribution.

The probabilistic approach is based upon statistical distributions. For each parameter, a series of measurements is carried out. These measurements are fitted to the most appropriate distribution (normal, log normal, uniform, etc.). The analysis is

usually carried out using a Monte-Carlo simulation, where random values are drawn from distributions. These random simulated values are run through the analysis, just as in the case of the discrete approach. Repeating this procedure a large number of times makes it possible to arrive at a conclusion despite the randomness and the uncertainty of the input distributions and the Monte Carlo procedure. The results for each run of the model are collected in the form of distributions that represent the final results. The probabilistic approach is often used to model complex systems where parameters are random in nature and easy to measure.

Monte-Carlo simulations can be only applied when sufficient statistical data are available for the estimation of distributions. In addition to this need, the probabilistic approach cannot overcome the problem of non-random uncertainties best represented by qualitative parameters (weathering of the rock mass, roughness of joints, etc.).

The approach that is proposed by this study is based on the fuzzy theory introduced by Zadeh [7]. This theory simulates the human ability to make decisions.

Non-random uncertainties could be analyzed through fuzzy systems and fuzzy sub-sets. These subsets can replace discrete numbers in the same way that statistical distributions can replace parameters.

Fuzzy calculations can be performed in several ways. Juang et al. [8] proposed a Monte-Carlo-like method of calculation that is similar to the classical Monte-Carlo simulation. Though it is very interesting, it could only be applied when fuzzy parameters are represented by Gaussian-like distributions.

The most widely used method of fuzzy calculation is the "α-cuts" method, where all fuzzy numbers are transformed into a number of intervals at the α level, and calculations are done on these intervals using interval mathematics. Figure 2 illustrates the discretization of a fuzzy number into four α-cuts

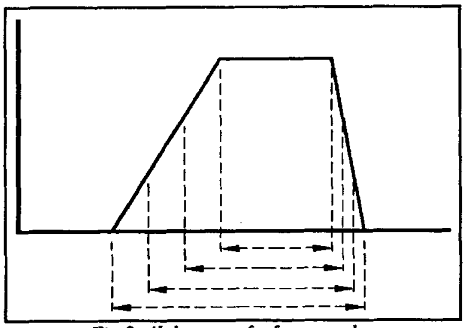

Fig 2. Alpha cuts of a fuzzy number

Fuzzy reasoning (sometimes referenced as approximate reasoning) is another point of research. Because of the nature and the properties of fuzzy numbers, we cannot perform a simple If-Then style of reasoning as presented by Eq. (1), because fuzzy reasoning consists of qualitative reasoning by statements as in Eq. (2).

$$\text{If } X = 3, \text{ Then } Y = 10 \quad (1)$$

If *Mechanical Fractures* are *Evolving*, Then *Risk* is *High* (2)

where *Mechanical Fractures* and *Risk* are fuzzy parameters, and *Evolving* and *High* are fuzzy subsets.

Fuzzy reasoning uses tables of rules that are defined by experts. Among fuzzy reasoning methods is the "min-max" procedure defined by Cox [9] and illustrated in figure 3.

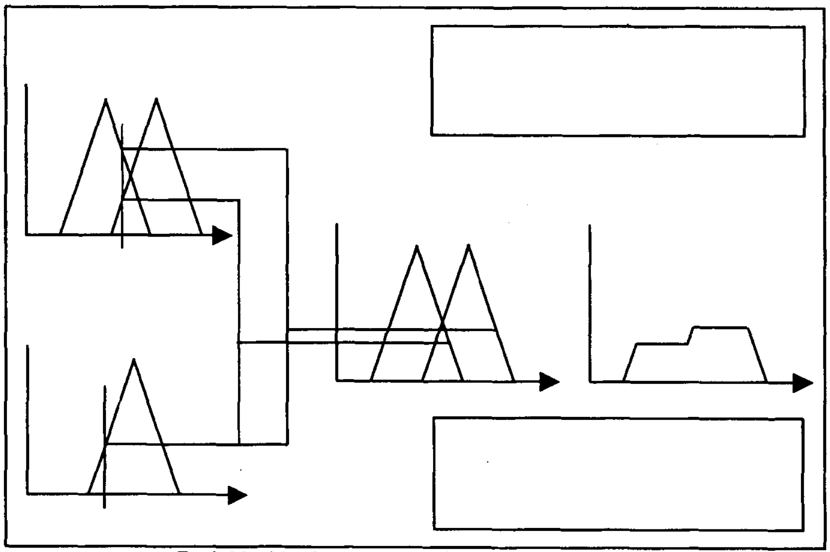

Fig 3. Min-Max fuzzy reasoning for a throttle action giving pressure and temperature as fuzzy numbers, after Cox [9]

3. General Methodology

In order to be able to define a general methodology for geotechnical risk analysis, we have combined many parameters into a general scheme presented in Figure 4. The parameters used in the analysis are usually defined as distributions for each specific site. We were interested in generalizing the choice of parameters in order to produce a more all-purpose methodology applicable to all sites and all cases. Therefore, the parameters that are used in our methodology are based on Bieniawski's [10]Rock Mass Rating (RMR) system classification for underground structures and Romana's [11] Slope Mass Rating (SMR) for slope and cliff structures.

The procedure for geotechnical risk analysis generally consists of four stages. The first stage consists of observation, where a site visit is required in order to collect

observations and information about the site; this may include parameters such as discontinuity spacing, weathering of the rock mass, etc.

The second stage of the analysis consists of using these parameters to calculate three main parameters, namely sensitivity, activity and intensity of the site.

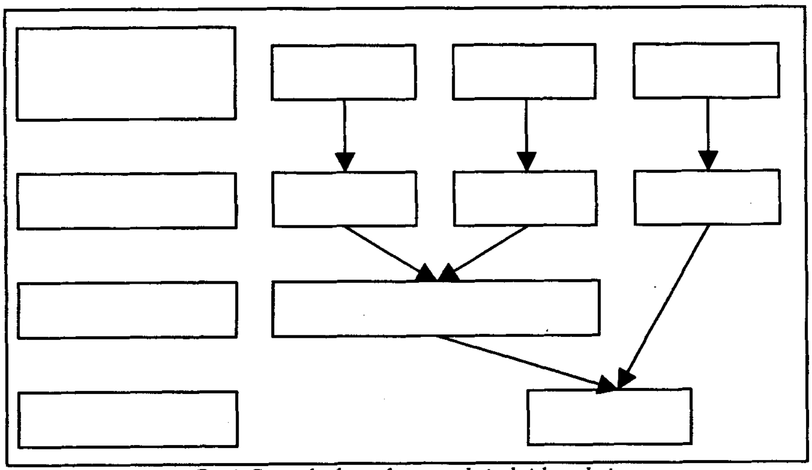

Fig 4. General scheme for geotechnical risk analysis

Sensitivity can be regarded as the potential capacity of the site to arrive at a condition that could generate an accident. In our case, the sensitivity is the result of the RMR or SMR evaluation. The *activity* of a site indicates the likelihood of site movements. It is defined using parameters like the opening of mechanical fractures and weathering of the rock mass.

In order to evaluate the severity of a possible accident, we observe also certain parameters that define the *intensity* of possible phenomena. Intensity is defined by calculating either the basic size unit of possible falling rocks, or the total volumetric size of these rocks.

In a discrete approach, for example, each observed parameter is quantified using an index and is then classified into a corresponding class; the same thing is done for the sensitivity, activity and intensity of the site.

The third phase of the analysis creates a matrix that contrasts the various classes of both sensitivity and activity in order to describe the possibility of activity occurrence. The fourth phase creates another matrix, using the different classes of possibility of occurrence (derived from phase three) and intensity to produce the final risk. Table 1 shows an example of a matrix describing the sensitivity and the activity of a certain site.

TABLE 1. Matrix for sensitivity and activity

Sensitivity (RMR) / ACTIVITY	Class I Very poor	Class II Poor	Class III Fair	Class IV Good	Class V Very good
SLEEPING	Negligible	Negligible	Low	Low	Interm. *
Inactive	Negligible	*Low*	Low	Interm. *	Interm. *
Fresh	Low	Low	Interm. *	High	High
Active	Interm. *	Interm. *	High	High	High

*** *Intermediate***

4. Geotechnical Risk Analysis

In the application of the above-described methodology to the analysis of geotechnical risk at a specific site, care has to be taken in dealing with problems of uncertainty. When using the discrete approach, three major problems are faced. First, the problem of variability that we have previously discussed. Second, the risk analysis must quantify qualitative parameters such as rock mass weathering; rock engineering experts often use linguistic variables such as "highly weathered" to describe the rock mass). The third problem is the "class limits" problem. This problem arises when the point estimate of an observed phenomenon falls near or at the limits of its corresponding class. This problem is illustrated in Figure 5, which shows a typical problem of class limits for RQD defined by Equation (3):

$$RQD = \frac{\sum Individual\ rock\ core\ lengths > 10cm}{Total\ core\ length} \qquad (3)$$

In contrast, the probabilistic approach can handle variability in parameter values, providing that sufficient statistical observations are available and that the corresponding statistical distributions are known. Although the problem of class limits could be considered as partially solved, the probabilistic approach is not capable of handling the qualitative parameters.

However, a major disadvantage of the probabilistic approach is the necessity of running a large number of simulations in order to arrive at satisfactory results.

Fuzzy logic, on the other hand, is capable of handling all three basic problems—advariability, representation of qualitative parameters, and class limits. These three advantages, in addition to the reduced need for computer power, encouraged us to adopt this approach for geotechnical risk analysis.

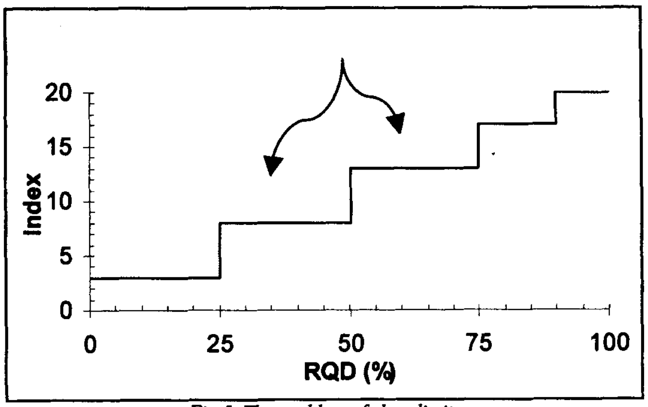

Fig 5. The problem of class limits

5. Applications and comparisons

We have applied the general methodology of geotechnical risk analysis to a case of an underground historical tomb at the Valley of the Kings, Egypt. The tomb is facing typical problems of unstability, showing possible risks of block movements, and requires a global risk analysis study. Figure 6 shows a plan of the tomb as well as the main discontinuities that were observed at the site.

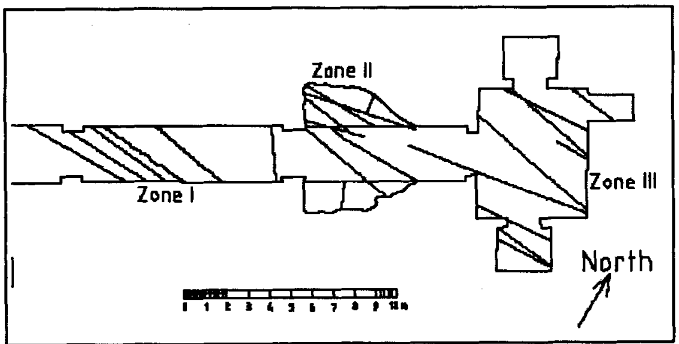

Fig 6. Plan of the tomb of Ramses I in the Valley of the Kings, Egypt

We have applied the rock mass classification system for the definition of sensitivity, as well as other parameters defining the activity and the intensity of block falling phenomena. A discrete approach analysis indicated low risk at zone III. Figures 7 and 8 show the results of the analysis at zone III using both probabilistic and fuzzy logic approaches.

We can see from these results that the fuzzy approach provides the most information about the risk state at the site. For example, the fuzzy analysis shows 71%,

24%, and 5% potentiality of low, intermediate, and high risk, respectively. The probabilistic approach gives probability of a high risk, while the discrete approach indicates only low risk.

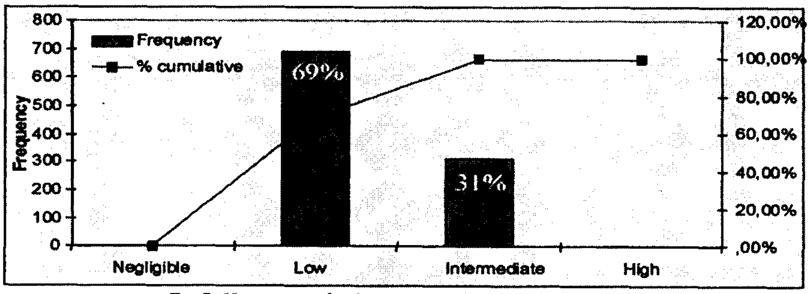

Fig 7. Histogram of risk at zone III (probabilistic approach)

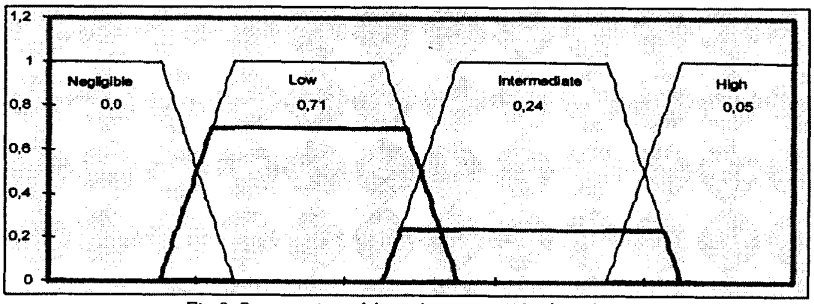

Fig 8. Presentation of the risk at zone III by fuzzy logic

6. Conclusions

Today, the analysis of geotechnical risks is a difficult work. As such, it must not be confused with deterministic stability analysis. A risk analysis study has to take into account uncertainty, or lack of information about the site, in one way or another. We have proposed a general methodology for geotechnical risk analysis that is based partly on the rock mass classification systems and that uses fuzzy parameters and fuzzy reasoning to better consideration of parameter uncertainties. We have demonstrated a typical example of the use of this method in an underground location, describing how the fuzzy methodology provides more information than a classical discrete approach or a probabilistic approach.

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COMPARATIVE RISK ASSESSMENT AND ENVIRONMENTAL IMPACT ASSESSMENT: SIMILARITY IN QUANTITATIVE METHODS

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Abstract

Management of municipal solid waste sites, toxic liquid waste sites, and former military and industrial contaminated sites is a pressing problem for most urban areas. In most cases these sites require management decisions when planning and instituting redevelopment. Very often decision-making is necessary for one of two cases: development of a new project on a contaminated site, or site conservation. Space limitations, health of local populations, preservation of historic sites – these urban area factors impose significant restrictions on the decision-making process. This present paper will discuss a quantitative analytical method that can be implemented for both Comparative Risk Assessment (CRA) and Environmental Impact Assessment (EIA) of development and reconstruction projects at urban contaminated sites. This method was implemented for several projects in St. Petersburg, Russia, including a CRA for a new underground development on a downtown city site contaminated by petroleum products.

1. Introduction

Assessment lists, the Leopold matrix, flow charts, combined maps assessments – these are some of the methods that are widely used for environmental assessment and decision-making. The method discussed in this paper is based on a joint analysis using two concepts: the natural-technical system, and sustainable development. A natural-technical system encompasses an artificial environment and ecosystem on the one hand, and a certain development goal on the other hand. Newly built developments, contaminated sites, landscape reconstruction – these are some examples of assessment goals. My method is based on a comparative multi-criteria assessment of quantitative and qualitative changes in the environment, and includes five steps. Figure 1 shows the general steps of the method.

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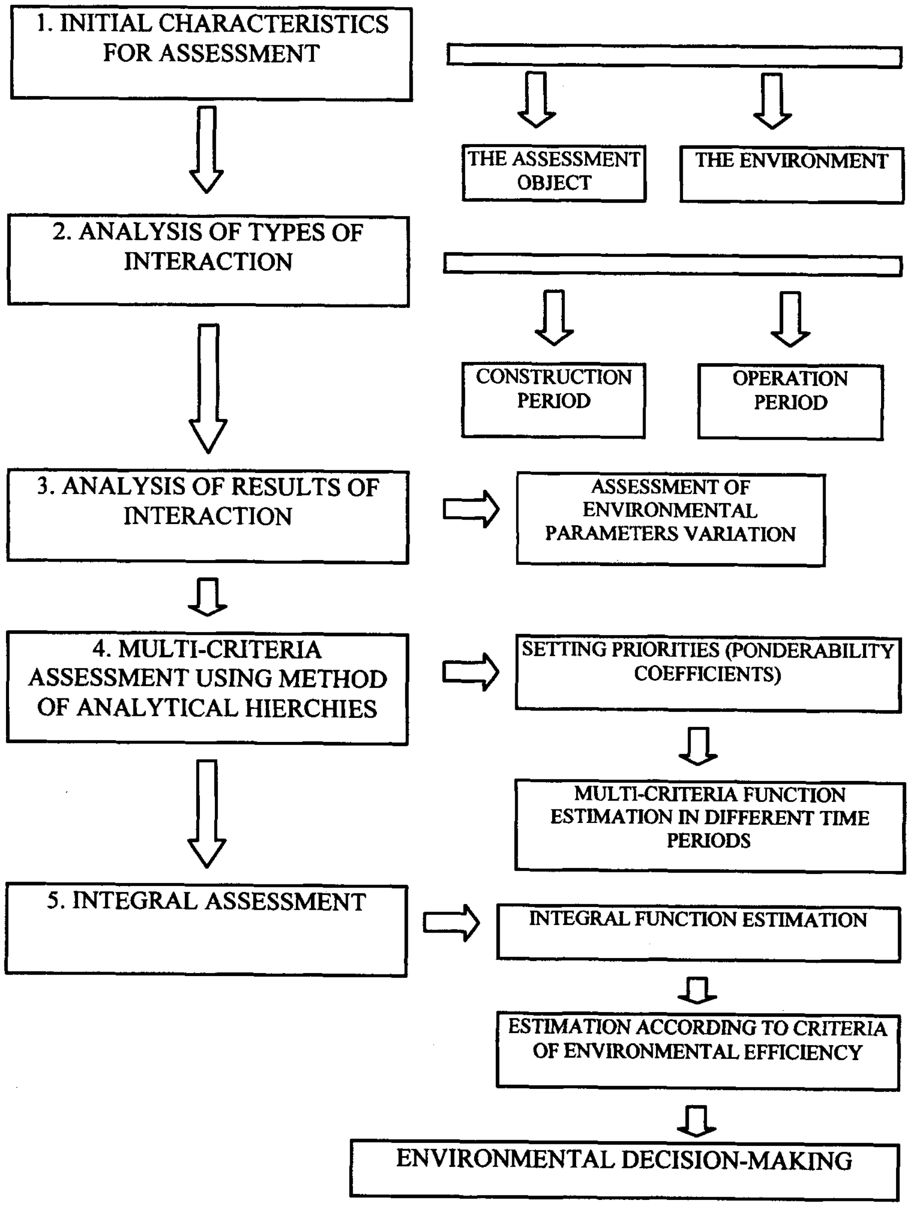

Fig. 1. General scheme of the method

2. Collection of initial information – data on environment and proposed engineering solutions

Since information about proposed actions and engineering solutions during project implementation is normally well defined, the important problem is to collect data on the environment. The environment encompasses biotic, anthropogenic, and technogenic factors. Collection of initial information on some of environmental characteristics may require complicated and expensive research activities (e.g. groundwater chemical composition analysis, transport streams surveying). Here are some examples of environmental terms and characteristics: geological, hydrological, landscape (including existing buildings), atmospheric, biotic, social. Assessment object and its environment also have some joint characteristics, e.g., ground stress-strain condition.

3. Analysis of consequences of proposed actions and risks

It is appropriate to consider two groups of environmental effects – those that might occur during the period of construction, those that might occur during the period of operation. Normally one would discover negative environmental effects during the construction period and positive environmental effects during the operation period.

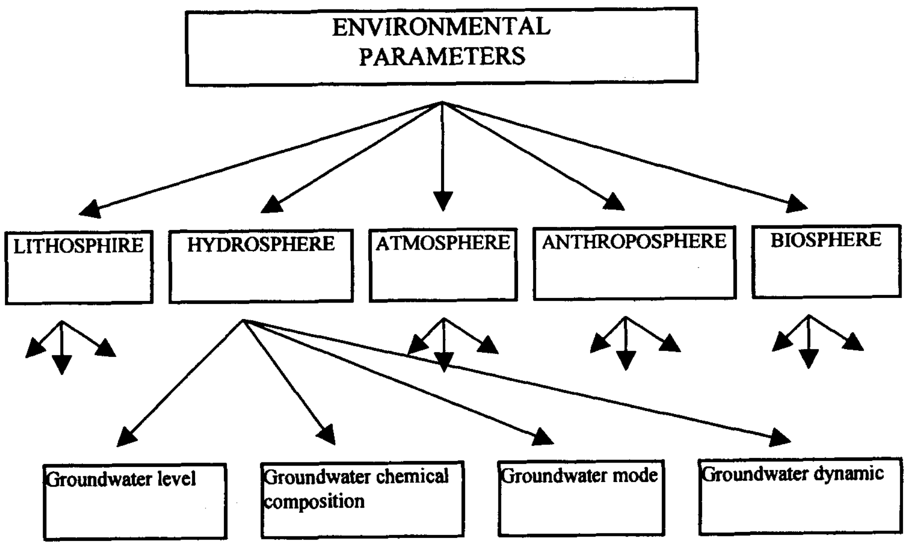

Fig. 2. Example of a hierarchy structure of environmental parameters

4. Environmental quality assessment and hierarchy structure of environmental parameters

I propose to assess environmental changes for development projects using a number of environmental parameters, which are composed in a hierarchical structure. This hierarchy resembles a tree, with upper level parameters summarizing lower-level parameters. This type of structure allows one to compare different environmental parameters situated on the same scale. For instance, to compare changes in groundwater level with changes in the chemical composition of atmospheric air, one should use their summarised parameters: hydrosphere and atmosphere.

The number of lower-level environmental parameters and the assessment of change or variation in each parameter depend upon the specific project. Figure 2 represents an example of a hierarchy structure of environmental parameters.

5. Multi-criteria assessment of the environmental parameters variation

The mathematical basis for the multi-criteria assessment of changes in environmental parameters is described by a method of analytical hierarchy process developed by Tomas Saati, who has also developed its implementation in his "Expert Choice" software. After developing a hierarchy structure, Saati's method requiring the following data inputs: pairwise comparisons of environmental parameters at lower levels with respect to the goal, pairwise comparisons of environmental parameters in groups – setting priorities and weights. For the implementation of Saati's method in EIA and CRA, it is useful to consider environmental quality as a goal for assessment.

Here is an example of an assessment of fluctuations in atmospheric air quality. The values used in Table 1 reflect the environmental conditions at Truda Square in St. Petersburg, Russia, during underground subway construction. The initial data describes the total output of pollutants in tons per year from the construction area during the following stages of the project implementation:

- Initial – N=97
- Construction – S=21
- Operation – R=44

TABLE 1. Example of comparison table arrangement

Total output of pollutants, tons/year	***N***	***S***	***R***
N=97	1	21/97=0,21	44/97=0,45
S=21	1/0,21	1	44/21=2,09
R=44	1/0,45	1/2,09	1

Resolving this comparison using Saati's method, we arrive at the multi-criteria function, *w*:

$$w = \begin{pmatrix} 0{,}124 \\ 0{,}592 \\ 0{,}284 \end{pmatrix}. \qquad (1)$$

The example given above illustrates the most favorable situation for such assessments, when the numerical data on changes in environmental parameters is available. In most cases, the pairwise comparison of changes in environmental parameters should be done by an expert in statistics. This, of cause, refers to setting up weights as well.

Results for this stage of assessment are given by values of multi-criteria weights for a given period of assessment. In the simplest case, the result is a priority graph that includes values describing the minimum three periods of assessment (initial, construction, operation).

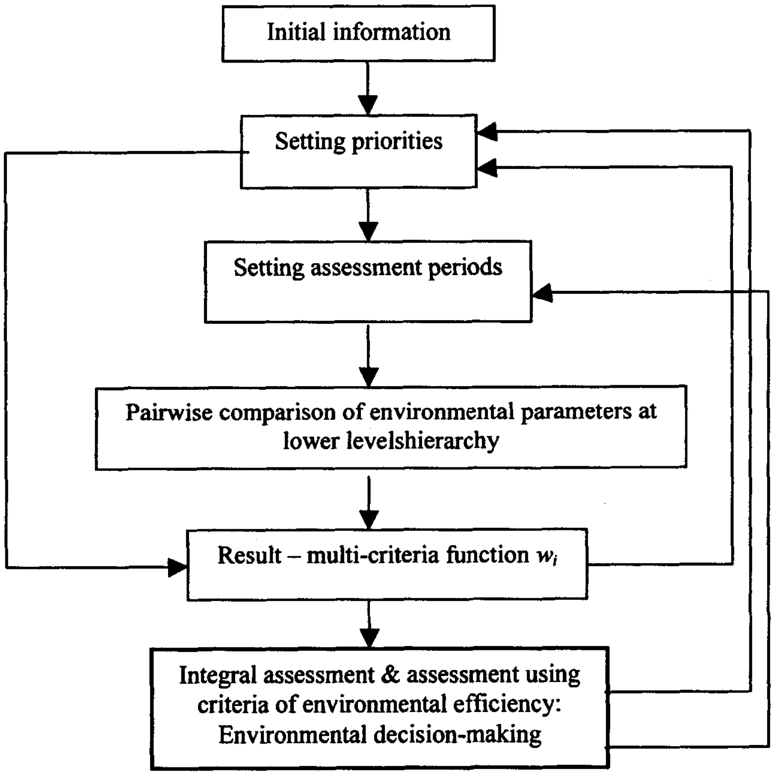

Fig. 3. Flow chart of the environmental assessment method

6. Integral assessment and environmental decision-making

The integral assessment is based on the values derived from Saati's method. It is proposed to use the following integral assessment function:

$$Q_j = \frac{\sum_{t=1}^{m} k_{jt} R_{jt} + \sum_{t=1}^{h} \bar{k}_{jt} (S_{jt} - N)}{N} \rightarrow \max, \qquad (2)$$

where:
Q_j – integral assessment value for an j project alternative,
t – duration of a particular assessment period,
m – number of assessment periods within the operation period,
h – number of assessment periods within the construction period,
k_t – time coefficients as follows:

$$k_t = \frac{t}{T}, \qquad (3)$$

where:
T – the whole period of project assessment.

Function 2 represents the ratio between sums of multi-criteria function values during all periods of the existence of an object being assessed and during initial environmental conditions before the project implementation. The integral assessment function allows comparison of calculations for different project alternatives, taking into account the entire lifecycle of the assessment object.

For environmental decision-making, two criteria of environmental efficiency are used:

$$Q_j > 1, \qquad (4)$$

$$\frac{S}{N} > 0{,}5 \cdot \qquad (5)$$

A flow chart describing the implementation of this method using the "Expert Choice" software, function 2, and criteria 4 and 5 is presented in Figure 3.

7. Practical implementation

The described method was implemented for EIA and CRA of several new building development projects in St. Petersburg, Russia, and Hamburg, Germany. Results of the assessments for a new underground development project on Konjushennaja Square, St. Petersburg, are given in Table 2.

TABLE 2. Results of an underground structure assessment at Konjushennaja Square, St. Petersburg.

№	*Alternative design solutions*	*N*	S_t	R_t	Q_i	*S/N*
1	4 levels; 423 car-parking capacity; retainer structure: diaphragm wall (inside), shpunt (outside).	0,373	0,215	0,412	<u>1,052</u>	<u>0,567</u>
2	3 levels; 282 motor-parking capacity; retainer structure: two rows of shpunt.	0,424	0,206	0,370	0,825	0,485

*Underlined figures fit criteria of environmental efficiency.

The most interesting peculiarities of the proposed underground project are related to complicated ground conditions at the site. Here is some data on groundwater at the site: the groundwater level is 2 m below the surface; it is contaminated by hydrocarbons (30 times more than the allowable limit, mercury (4 times more than the allowable limit), and organic compounds. Approximately two-thirds of the site is supposed to be replaced.

8. Conclusions

This proposed quantitative analytical method for CRA and EIA enables assessment of the environmental effects of a project, and decision-making regarding project implementation from an environmental standpoint. The method can be used as a line of evidence for estimating the overall effects of a project, involving political, social, economic, etc. criteria. The method can be successfully implemented for projects with negative environmental effects, as well as for those having beneficial ones. In the latter case, when no environmental deterioration is to be expected, it is possible to select a project having the most beneficial environmental impacts (lowest risk), or make decisions on the basis of non-environmental criteria.

This method can be used for a wide range of projects, especially for civil engineering projects.

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COMBINING EXPERT JUDGEMENT AND STAKEHOLDER VALUES WITH PROMETHEE: A CASE STUDY IN CONTAMINATED SEDIMENTS MANAGEMENT

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Abstract

Management of dredged contaminated sediments can be a contentious, difficult, and expensive task. Because the waterways from which sediments are dredged have multiple uses, competing interests are often brought to bear on any decision. No single best alternative is likely to emerge; different stakeholder groups will prefer different alternatives. This chapter investigates the utility of multicriteria decision analysis (MCDA) as a tool for incorporating stakeholder values into the decision process, for soliciting public participation, and analyzing novel technological alternatives. An outranking method called PROMETHEE is employed for three reasons. First, the emphasis placed on assessing new technologies – and especially beneficial reuse technologies – requires a method that facilitates introduction of new alternatives at any point during the analysis. Second, outranking methods are conducive to elucidating the contrasting value structures of different stakeholder groups. Third, they are more capable of handling semiquantitative scales (e.g., high, middle, low) than optimization methods such as MAUT or AHP. To illustrate the decision process under development, this chapter presents the results of a case study example involving stakeholders in Dover, New Hampshire concerned with the dredging of the Cocheco River.

1. Introduction

Millions of cubic yards of contaminated sediment are dredged from navigable waterways every year (USACE 2003) and management of these materials can often be a difficult and contentious task. New restrictions on ocean dumping of contaminated sediments have created a need to explore other, potentially more expensive alternatives, including those that create beneficial reuse opportunities. However, like all environmental problems, dredged sediments management involves shared resources (such as waterways). Engaged public and private stakeholder groups often hold

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competing views or priorities, including environmental protection, jobs or economic opportunity, protection of cultural or religious traditions, and environmental justice. Consequently, the parties most interested in or impacted by any environmental project may not be able to agree on a common set of priorities or goals. Moreover, due to increasing complexities and different perspectives, the scientific experts and stakeholder groups involved can often feel disconnected from each other in the decision process. Partly because resources are scarce, and partly because some goals may be mutually exclusive, not all goals held by all stakeholders can be satisfied in every instance.

Nevertheless, early involvement of public, non-expert stakeholder groups may significantly improve environmental decision-making processes, including policy making, or environmental design (Gregory & Keeney 1994; National Research Council 1996, Corburn 2002, Renn et. al. 1995). Stakeholder input may come in the form of helping to set the decision context, specifying objectives to be achieved, identifying alternatives, and incorporating non-expert knowledge. There is also support for making more constructed and integrated decisions by focusing on stakeholder *values* (Keeney 1992, EPA 2000). While there is a body of literature on varying methods to incorporate stakeholder values into environmental decision-making (e.g. Dale & English 1999, Gregory & Wellman 2001, Gregory & Keeney 1994, McDaniel & Roessler 1998, Wilson & Howarth 2002), the concept and practice are still novel and evolving.

Multicriteria decision analysis (MCDA) is one way of balancing the competing objectives of different groups vested in an environmental problem. An innovative branch of decision analysis, MCDA is as an all-encompassing term that defines the processes used to help individuals or groups examine the multiple considerations that go into decision making while identifying the various trade-offs, conflicts and potential coalitions that exist within the decision context (Belton & Steward 2002). In an MCDA approach, a decision may be understood from multiple perspectives. The advantage of MCDA (over simple, single objective optimization problems) is that MCDA can take into consideration multiple objectives and compare alternatives in many different ways, as opposed to trying to reach one ultimate solution. The purpose of MCDA is to clarify the complex decision problems, not optimally solve for the problem. MCDA compares alternatives based on incommensurable criteria and allows conflicting components of the decision problem to be examined simultaneously (Hermans 2003).

Selecting from among the many MCDA methods available may be an arduous task in and of itself. This project employs an outranking procedure called PROMETHEE (Preference Ranking Organization METHod for Enrichment Evaluations) that compares the performance of different technological alternatives on the basis of decision criteria selected by stakeholders. Unlike optimization methods such as multiattribute utility theory (MAUT) or the analytical hierarchy process (AHP)), outranking methods are not focused on synthesizing multiple criteria into a single recommendation. The purpose of PROMETHEE in this context is to foster mutual understanding among stakeholders with different viewpoints and the decision maker(s). The methods are described elsewhere in the book (see Chapter 1) and the strengths and weaknesses need not be reiterated here. It is sufficient to say that the principal reason for using outranking methods is that it facilitates the separation of

stakeholder value elicitation from performance analysis – allowing introduction of new alternatives at any point in the decision process. This is essential in situations like the case study discussed below in which new or evolving technologies are being introduced. Less flexible methods that require all alternatives to be fully defined ahead of time could be too costly or time consuming to implement under conditions in which new technologies are brought to bear. However, this research hypothesizes that stakeholder values and decision criteria remain consistent (or change slowly) over the course of a project, regardless of technological developments, and that new alternatives can therefore be assessed (at least in a preliminary fashion) with regard to separately established stakeholder preferences.[1] In conjunction with this is the hypothesis that understanding stakeholder values may allow technical experts to predict individual preferences for management alternatives, enabling managers and engineers to prioritize research into development of the most promising new alternatives.

2. Case Study Background

To partially test these hypotheses, this work studied a proposed contaminated sediment management project involving the Cocheco River in Dover, New Hampshire. The Cocheco River is located in the southeastern part of New Hampshire and flows toward the Gulf of Maine and the Atlantic Ocean. "The main stem of the river drops approximately 900 feet vertically from its highest elevations at Parker Mountain in Strafford and Birch Ridge in New Durham to the wide sections impounded by dams in the lower river. Below the dam in Dover, the Cocheco is a tidal river."[2] The proposed section of the river to be dredged is the part just below the dam in the center of the city of Dover to its confluence with the Piscataqua River. Approximately 45,000-60,000 cubic yards of sediment, some of which are contaminated with polyaromatic hydrocarbon (PAHs) and heavy metals, are planned for removal.

There are many motivations for the dredging project -- including maintenance of a navigable channel – which is considered essential to the long-term economic development plans to return the City to its former status in the 19th century as an inland port. Because it is a navigable waterway, the U.S. Army Corps of Engineers (USACE) has been helping the city of Dover coordinate the process and will be performing the dredging.

A brief environmental history of the channel demonstrates that the sources of contamination and events leading up to the current dredging project may be typical of dozens of small New England cities.

- In the **mid-to-late 1800s** the channel allowed three-masted schooners more 100 feet long to call at Dover's inland Port.

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[1] There certainly may be situations in which this assumption proves fallacious, such as the introduction of radically new technologies (e.g., genetic engineering) that raise questions or engender potential environmental effects never contemplated before. Exempting radical changes, the outranking process allows an iterative exchange between stakeholder value elicitation, technology assessment and further development, and verification or reelicitation that is more consistent with a model of decision *evolution*, rather than decision making at a single moment in time.

[2] http://www.state.nh.us/coastal/coast/cocheco.htm

- In **1896** a large storm filled in the river, blocking channel access to all but small boats.
- From **1910-1950** a riparian site (subsequently used for disposal of contaminated sediments and currently occupied in part by the municipal wastewater treatment works) hosted a waste incinerator.
- In the **1960s** the site became home to the city landfill, which remained unlined and uncapped under 1981 law that exempted existing landfills from closure requirements.
- During **1985** a small part of the river was dredged. Contaminated spoils from this dredge were buried at the former landfill site, which was eventually graded and developed as a recreation area known as Maglaras Park.
- In **1996** the U.S. Senate approved a bill authorizing $600,000 to fund a larger dredging project. Testing of in-situ contaminant levels began.
- In **May 1997** the Dover City Council approves $40,000 in capital funding.
- By **July 1997** environmental testing revealed elevated metal and industrial organic pollutant concentrations in along the bottom of the river, eliminating ocean dumping as a viable disposal alternative. The local newspaper reported that "Lab workers testing Cocheco sediment for the Army Corps in the 1990s noted they could smell hydrocarbons in most samples. They were told not special precautions were needed but only one of the 14 samples did not have to be disposed of as hazardous waste" (Emro 2002).
- On **August 10, 2000** the Waste Management Division of the New Hampshire Department of Environmental Services (NHDES) granted a waiver allowing the city to construct a dredge spoil cell for the Cocheco River Maintenance Dredging project on the former landfill site located near the north end of the area to be dredged.[3] The city needed this wavier because testing of the sediments in the river had shown that there were contaminants such as the heavy metal chromium and PAHs that were in excess of requirements for an unsecured landfill. As all other alternatives for disposal had been reviewed and discarded for a number of reasons (e.g., feasibility or expense), this waiver was the key element in the continuance of the dredging process.
- During **2001** a local citizen's group prompted the city to test the soil at the proposed disposal site (and current recreation area). Elevated levels of PAHs polyaromatic hydrocarbons (PAHs) were detected and attributed to the spoils from the 1985 dredging, which include sediments that had been contaminated decades prior by an upstream coal gasification plant. The recreation fields were closed. (Emro 2002).
- **July 17, 2002** – NHDES approved dredging of 2.7 miles and removal of 45,000 cubic yards of sediment (65,000 cubic yards is the estimate when over dredging is taken into consideration) from the Cocheco River above its confluence with the Piscataqua River.

There has been and still is debate in the community over the dredging decision. However, the focus of this research is just on *disposal* of the contaminated

[3] NHDES Wetland Permit Application File #2001-932

sediments, rather than on dredging operations themselves. In this regard the actual decision process employed by City officials was unstructured compared to the MCDA approaches described in other sections of this book. Regulatory constraints required secure disposal of contaminated materials (i.e., prohibiting ocean dumping). The closest authorized landfill, the privately-owned Turnkey landfill in Rochester NH, is only 12 miles from Dover, but refused to accept dredge spoils. The next nearest landfill (in Maine) was prohibitively expensive due to the transportation costs and tipping fees, so secure landfill disposal was judged to be infeasible.

The next most secure option appeared to be a constructed upland disposal site. The City looked at nearly a dozen sites in close proximity to the dredging operation, but ruled out those that were presently undisturbed or unsuitable for dewatering of dredged spoils (i.e. too steeply sloped). Officials felt that priority consideration should be given to sites that have already been compromised, such as the former landfill and abandoned recreation area. The potential to use the dredged material and a clean cap to cover the contamination already placed at the former landfill site was attractive to City decision-makers because it seemed to offer the potential to improve an undesirable environmental condition and perhaps reclaim the recreation area. Nonetheless, this site required application for a waiver from NHDES.

During the process of completing dredge permit and disposal cell waiver applications, there was opportunity for public involvement in two ways: public comment on written application materials, and public hearings. Three separate applications could have initiated public comment or hearings: the USACE dredge proposal, the waiver application, and a wetlands permit application. The first two generated insufficient public comment to warrant a hearing, even though the City was required to notify abutters and plans were publicly available. However, the wetlands permit application generated significant interest. A number of questions regarding the long-term viability of the site were raised, but ultimately the site was presented as the only feasible disposal alternative, and the waiver was granted.[4] Table 2 summarizes the alternatives that were considered by the City of Dover and USACE along with the various factors that were taken into account in this determination.

As part of the waiver approval agreement, the City is obliged to pay for and tend to the monitoring and upkeep of the disposal cell. USACE is currently putting together an Operations and Maintenance Plan for the cell, which is also part of the final approval process. The cell will be left uncapped for one year while the liquid settles out. A fence will be placed around the cell during this time to reduce the risk of public harm. The dewatering liquid will travel to the Dover Waste Water Treatment System, which according to hearing materials in the file, "confines the contaminants to a manageable location."[5] "The secure dredge spoil dewatering area (DSDA) will be lined to collect sediment-dewatering effluent and provided with an impervious cover to prevent infiltration and potential direct contact with the dredged material."[6]

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[4] Public Comment. NHDES Wetland Permit Application File #2001-932

[5] NHDES Wetland Permit Application File #2001-932

[6] Sills, Michael. Waiver Request for Cocheco River Maintenance. NHDES. July 20, 2000.

TABLE 1: Feasibility Study of Disposal Alternatives

Alternative	Considerations
Turnkey Landfill	Refused to accept because of volume and characteristics of sediments.
Ocean Dumping	Unacceptable because of contaminants.
Upland Disposal Sites Along the River	Land was undisturbed (in natural state) or unsuitable (e.g. grades too steep)
Secure landfill site in Maine	Transportation costs were too high
Local Landfill Remediation (Superfund Site)	Contaminants were not suitable for this process
Former Landfill Site / Dover Public Works	Costs of upkeep and monitoring were acceptable Proximity to river minimizes transportation risks. Officials say there will minimal environmental effects. Others are skeptical. Waiver needed to build disposal cell

The construction of the cell will be inspected by USACE and the City with the city of Dover becoming the final owner of the site. Some of the disposal area is part of Maglaras Park where PAHs from the disposal of the 1985 dredging were discovered on the soccer and baseball fields.[7] There are future plans to use the new spoil site as fields again, once the cell has been capped and covered and if safety standards allow.

3. Structuring an MCDA, Value-Focused Approach

The decision making process employed by local authorities in the Cocheco project may appear rational, practical, well thought out, and may have resulted in a wise choice. However, the introduction of new alternatives-- such as beneficial reuse technologies -- would have increased the complexity of the choices so significantly that it could have overwhelmed the simplified, heuristic decision process employed. (See Chapter 1). The principal objective of this study was to investigate methods of synthesizing MCDA tools with time and cost effective stakeholder participation methods that could speed the introduction of beneficial reuse technologies and stakeholder-based processes into smaller, heavily resource constrained communities exemplified by the City of Dover. There are a number of methods of stakeholder value elicitation and public participation available to choose from. These include public value forums (Keeney 1990), various types of surveys such as decision pathway (Gregory et al., 1997), contingent valuation (Gregory and Wellman, 2001) or multiattribute value integration (Gregory, 2000) surveys, stakeholder workshops (Gregory & Keeney 1994, McDaniels & Roessler 1998), and combination workshop and scientific model building (Borsuk et al., 2001).

The actual methods employed here rely upon a combination of the strengths and weaknesses of these, considering the resource limitations of the case. The study was designed to have three primary points of contact with stakeholders: a reflective conversational interview to establish the primary concerns, a written survey, and a verification interview. Following the initial interviews, the information regarding

[7] NHDES Wetland Permit Application File #2001-932

stakeholder values was forwarded to experts for additional characterization of the decision criteria (by identifying attributes of each, and if possible, appropriate metrics for these attributes) and assessment of the performance of the new technologies in the areas identified. Figure 1 shows a schematic of the methods employed, while details are described in the paragraphs below.

3.1 STRUCTURING THE DECISION AND IDENTIFYING KEY STAKEHOLDERS

Research began by investigating the background information available on the local case study. Business interests, government officials, local citizens and environmental advocacy group leaders and members were identified in public documents generated during the permitting and public hearing process. Additionally abutters to a proposed disposal site were identified, as were employees at the local wastewater treatment plan, individuals mentioned in newspaper articles, and anyone else recommended to researchers. The stakeholders fell into general four categories as detailed in the box at right.

Key Stakeholder Groups

Citizen/Environmental Advocacy Groups
Conservation Law Foundation, Save Dover, Cocheco River Watershed Coalition, Dover Conservation Commission

Business Interests
Greater Dover Chamber of Commerce, George's Marina

State & Local Government/State Agencies
City of Dover Environmental Projects Manager, NHDES, Dover City Councilors

Local Citizens/Abutters to the project
Individuals identified in public comment documents, or solicited from knowledge of proximity to affected areas.

3.2 INITIAL INTERVIEWS

Using an open-ended, reflective conversational format, representatives from each stakeholder group were interviewed personally or on the phone to identify key decision criteria and project objectives. Among the salient findings were four recurring themes: economics, environmental quality, human habitat, and ecological habitat. Although stakeholders differed in emphasis, each of these qualities was mentioned during many of the interviews. At this stage, stakeholders helped to characterize the major decision criteria by discussing how they could be measured or manifested in specific attributes. For example, economics was identified as an important decision criterion, but economic considerations have many facets differing in importance to different stakeholders. Project costs (80% of which are slated to be paid from Federal sources), maintenance costs, and community economic development (e.g., jobs) all were identified as driving the overall economic assessment.

3.3 THE WRITTEN SURVEY

The four major criteria were combined with expert input to create a written, value-elicitation survey that asked stakeholders to rate, rank, and tradeoff among the

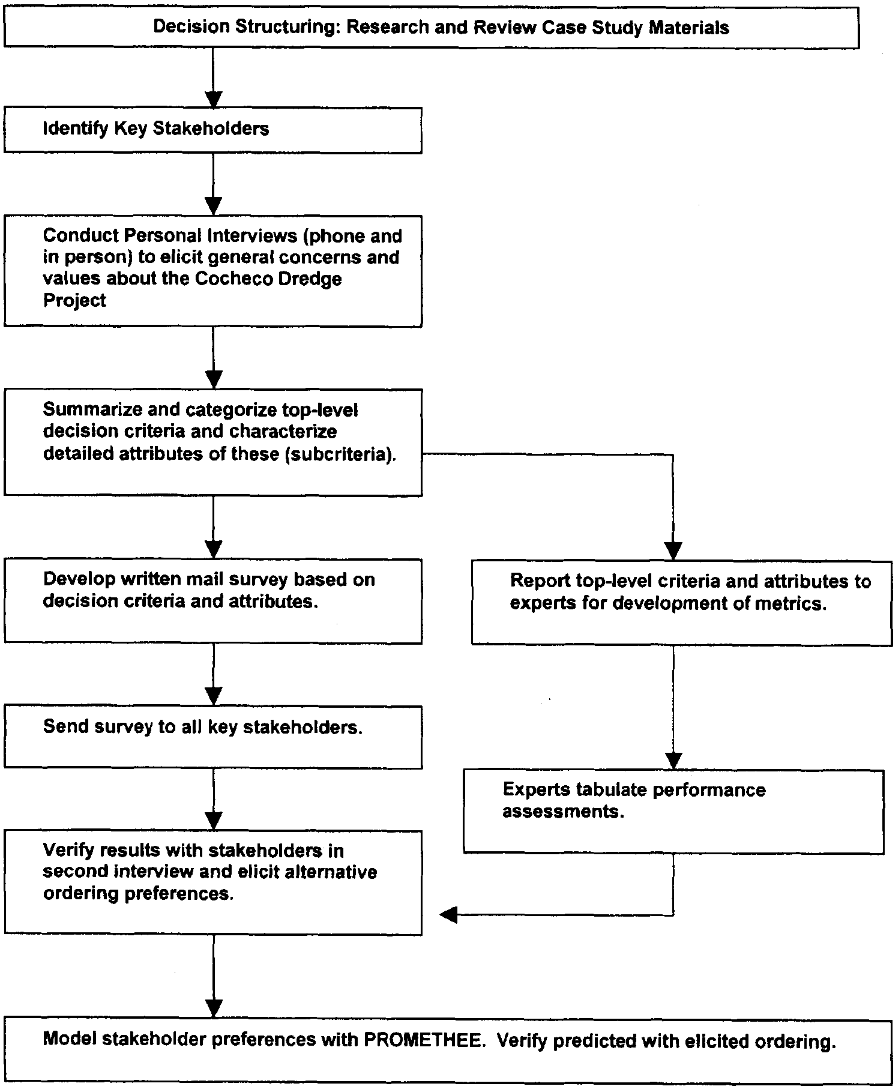


Figure 1: Stakeholder value elicitation and MCDA method.

interrelated criteria pertaining to the management of contaminated sediments. Several attributes of the top-level criteria were identified as sub-decision criteria:

flood plain/shoreline	treatment costs
building/infrastructure/commerce	capital costs
plant, animal & fish health	maintenance costs
water & air quality	tax revenues
recreation/open space	future risk of economic loss
human health	

Stakeholders were careful to point out that these attributes were interrelated rather than independent. For example, a single attribute such as preservation of floodplain and shoreline was perceived as being meaningful attribute of ecological and human habitat by most stakeholders, but also perceived by some as relating to environmental quality and economics. Figure 2 demonstrates how the decision criteria are interrelated. In particular, several attributes identified with "environmental quality" were also identified as relating to other decision criteria.

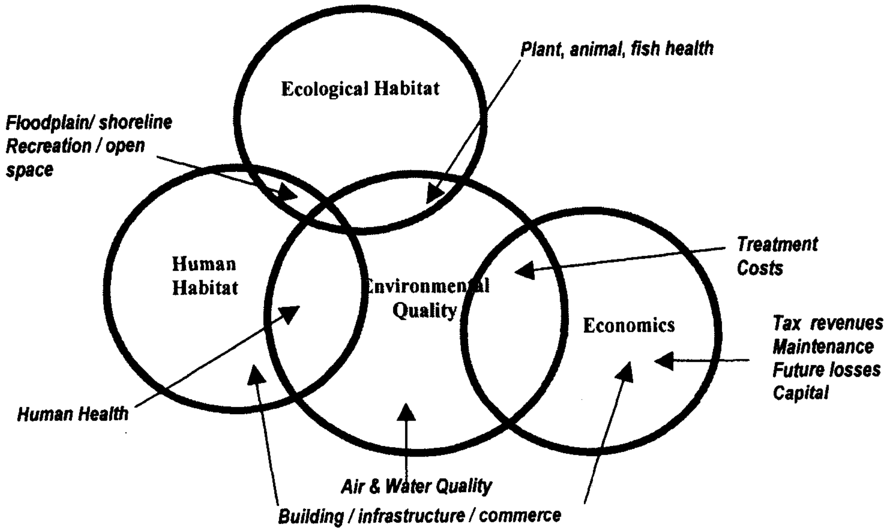


Figure 2: Attributes were perceived as being interrelated and meaningful to multiple decision criteria.

3.4 INTERPRET SURVEY RESULTS

The survey was sent to 15 key stakeholders and served as a tool to qualitatively and quantitatively measure their values. Stakeholders were asked to assign percentage weights for each of the four major decision criteria, and to rank all attributes in order of importance in several different groupings of four to eight. Attributes were compared

pair-wise and ranked to determine the dominance of some over others. When one attribute was consistently preferred to another attribute in all groupings, the preferred attribute was awarded a 'win point.' Intransitivites (in which one attribute may be preferred over another on two questions, but the order reversed in a third) were handled by awarding a partial win point (such as two-thirds for the first alternative and one-third for the second in the parenthetical example). A final ranking from 1 (most important) to 11 (least important) was established on the basis of the win points. In each case, respondent profiles emerged from the attribute rankings that were consistent with the criteria weightings. Both final attribute rankings and criteria weightings were verified with respondents in follow-up interviews.

3.5 EXPERTS TABULATE ALTERNATIVES PERFORMANCE TABLE

Graduate students and faculty at the UNH Center for Contaminated Sediments Research (CCSR) were interviewed about their research on beneficial reuse alternatives, which include wetlands restoration, cement manufacture, and flowable concrete fill. Interviews served as a forum to elicit assessments of each alternative to create a performance table that compares the attributes of each alternative to the others based on the four criteria identified by stakeholders in initial interviews. The results are summarized in Table 2

Table 2: Expert Assessment of Alternative Performance

Alternative	Cost ($/cy)	Environmental Quality	Ecological Habitat (acres)	Human Habitat (acres)
Cement Manufacture	$30 *+3.0*	High *+2.0*	0 *-1.0*	0 *-1.0*
Flowable Fill	$55 *+1.0, -2.0*	Medium *-2.0*	0 *-1.0*	0 *-1.0*
Wetlands Restoration	$75 *-1.0*	High *+2.0*	10 acres *+3.0*	0 *-1.0*
Upland Disposal Cell	$40 *+2.0, -1.0*	Medium *-2.0*	0 *-1.0*	4 *+3.0*

Notes: Expert assessment determined the performance of each alternative on the four salient criteria that stakeholders identified as important. The actual alternative planned for use in the Cocheco River Project is the Upland Disposal Cell. Dominance rankings are given in italics according to the number of clearly inferior (positive) or superior (negative) alternatives.

By combining the stakeholder-identified criteria with the expert performance information, one can see where each alternative dominates and is dominated by the others. For example, experts expect cement manufacture to be the least expensive – consequently outranking all three other alternatives with respect to cost. Moreover, cement manufacture is tied with wetlands restoration for the highest environmental quality assessment, with both outranking the two other alternatives. Wetlands restoration dominates all the others in creation of ecological habitat, whereas the upland

capped cell (that is planned for use in the actual project) dominates in human habitat – chiefly due to the plan to restore use of the recreation areas following capping of the cell. Moreover, it is clear from this table that flowable concrete fill is inferior to one or more alternatives in all respects. Based solely on this table, there would be no reason to choose this option in this case.

It is essential to recognize that the performance table represents the best available expert judgment of the performance of each alternative with respect to the criteria identified by stakeholders. Ideally, both experts and stakeholders would participate in devising the proper metrics to assess performance. The situation in this case study, however, was less than ideal. For the most part, the technologies under research are not developed fully enough to create reliable assessments or fully quantitative scales. For example, risk-based measures may be an improvement on a "high, medium, low" environmental scale; or assessed property values may be an improvement on acres of human habitat. In practice, any number of metrics may be proposed or devised, although many of them may not be measurable. Therefore, establishing the performance table may best be carried out as an iterative process engaging a back-and-forth between stakeholder's assessment of relevancy and expert assessment of measurability. However, once established, the performance table must not be in dispute. That is, stakeholders (or experts) may disagree on the relative importance of different decision criteria, but not on the assessment of the performance of any alternative with respect to the criteria identified. The performance table is intended to represent the facts as they can best be presented and understood, whereas the decision analysis allows introduction of opinion. Nevertheless, in this particular case, the performance assessments must be interpreted as hypothetical – both because the technologies themselves are not fully developed and because the details of the case study are not sufficiently developed to allow detailed assessments. To guard against bias among the stakeholders (in the absence of a full vetting of the details of each alternative) the title of each alternative was presented to stakeholders as Option #1, Option #2, etc., and the respondents were asked to evaluate each alternative upon the merits presented in the table. Also, the performance *rankings* (+3.0, -1.0, etc.) were not included in the performance table made available to stakeholders.

3.6 VERIFICATION PROCESS

Interpreted survey results were verified in a personal one-on-one interview with each stakeholder that completed the survey and was willing to participate in the interview (12 total). During this interview, stakeholders were presented with the 'win points' attribute rankings based upon analysis of the written surveys and asked to confirm and/or revise their interpreted values as well as elaborate on their concerns pertaining to the disposal of contaminated dredged sediment in general and in particular with the Cocheco Case. They were also asked to comment on the survey method as a tool to increase public participation in the decision process. Many of the stakeholders were critical of the actual survey tool, stating that it was hard to differentiate among the interrelated criteria and that they were not used to expressing their values about this type of decision. However, the majority agreed that researchers had faithfully captured their values in general and were willing to elaborate on them during the interviews. All

participants were agreeable to sharing their views and appreciated being incorporated into the research process. Additionally, the interviews served as an opportunity to begin eliciting stakeholder input on the beneficial reuse technologies. At the end of each interview, stakeholders were presented with the performance table and asked to make blind rankings of the four technological options.

3.7 PREDICTING STAKEHOLDER PREFERENCES FOR MANAGEMENT ALTERNATIVES

Verified survey results are used to create predicted management alternative rankings by utilizing the PROMETHEE method of pair-wise comparison embodied in the *Decision Lab 2000* software (Visual Decision Inc. 2000). In PROMETHEE, which stands for Preference Ranking Organisation METHod of Enrichment Evaluation, alternatives are compared and ranked under each decision criteria. Dominated alternatives (i.e. inferior in every respect) can be identified easily and trade-offs identified among the remaining alternatives highlighted. Individual stakeholder orderings are established by weighting decision criteria as a percentage of the overall decision in a manner consistent with expressed stakeholder values. Dissimilar individual stakeholder preferences are never summed or averaged. Each stakeholder may have a different ordering of the preferred alternatives. Consequently, PROMETHEE is especially useful for calling attention to potential conflicts or alliances between different stakeholder groups. (For more information on PROMETHEE, see Brans and Mareschal 1994 or Brans and Vincke 1985).

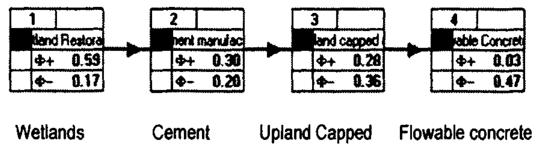
In PROMETHEE, rankings are based upon calculation of positive and negative 'flows', which are measures of the weighted average ranking of each alternative according to the performance table. For example, in an equal-weighting (or balanced) scenario, the positive flow for cement manufacture is calculated as the sum of the positive rankings +3.0 (from economics), +2.0 (from environmental quality), zero, and zero (from both human and ecological habitat), divided by the total number of spaces in the matrix made up of competing alternatives (in the rows) and criteria (columns), which is 12. The result is 5 divided by 12, or 0.42. Negative flows are computed on the basis of negative rankings. Lastly, overall alternative orderings may be determined by comparison of positive flows, negatives flows, or the sum of these. Often, the alternative orderings provided by the positive and negative flows are identical. When they are not, PROMETHEE may have identified alternatives that are incomparable. In this case, one alternative may exist that has both outstanding strengths and serious shortcomings.

Selecting this alternative may reflect a strongly held preference for the criteria assessed as strengths – a position that may generate controversy. An example ordering is shown in Figure 3, with positive flows reported in a small box above negative flows. Of the seven stakeholders that participated in the ordering of preferred alternatives, the decision analysis correctly predicted the elicited ordering of all four alternatives for three of the stakeholders (using the verified percentage criteria weightings supplied in the written survey). In the other four cases, the stakeholders' first and second choices matched exactly. These results suggest that the researchers can rely upon the stakeholder value elicitation instruments to communicate a reasonably well quantified

expression of values that can be employed to prioritize development of the current alternatives, or screen new alternatives that may be introduced into the decision process later. Moreover, while the decision matrix in this case was fairly simple for stakeholders to analyze heuristically, the consistency between predicted and elicited results suggests that the decision analysis may be a valuable tool to assist decision-makers in evaluating more complex situations in a manner consistent with stakeholder values.

Example Alternative Orderings

Predicted



Elicited

1. wetlands
2. cement
3. flowable concrete fill
4. upland capped

Figure 3. Based on individual preference functions, *Decision Lab* can predict the order in which any stakeholder would prefer available alternatives using PROMETHEE. Predicted results for all stakeholders were compared to the actual, ordering of alternatives elicited from stakeholder inspection of the performance table given to stakeholders during the verification process.

Researchers were also interested in the stability of the alternative orderings – i.e., the sensitivity of the orderings to the criteria weightings – and in the potential conflicts or alliances among different stakeholders groups. Both of these are readily investigated in PROMETHEE, as presented in *Decision Lab 2000.* For these purposes, stakeholders were grouped into one of four classifications that generally represented their most strongly held views:

1) The **Ecological Habitat and Environmental Quality Concerns** (Figure 4) group included those that were concerned largely with plant, animal and fish health, as well as the status of the environment particularly in terms of air and water quality. The order predicted for this group was (with positive and negative flows in parantheses):
   a. Wetlands Restoration (+0.60, -0.17)
   b. Cement Manufacture (+0.30, -0.20)
   c. Upland Capped Cell (+0.27, -0.37)
   d. Flowable Concrete Fill (+0.03, -0.47)

2) The **Balanced** (Figure 5) group included those who stated their concerns were equally weighted among all four criteria: economics, environmental quality, ecological habitat, and human habitat, and was predicted to prefer:
   a. Cement Manufacture (+0.42, -0.17)
   b. (tie) Wetlands Restoration (+0.42, -0.33)
   c. (tie) Upland Capped Cell (+0.42, -0.33)
   d. Flowable Concrete Fill (+0.08, -0.50)

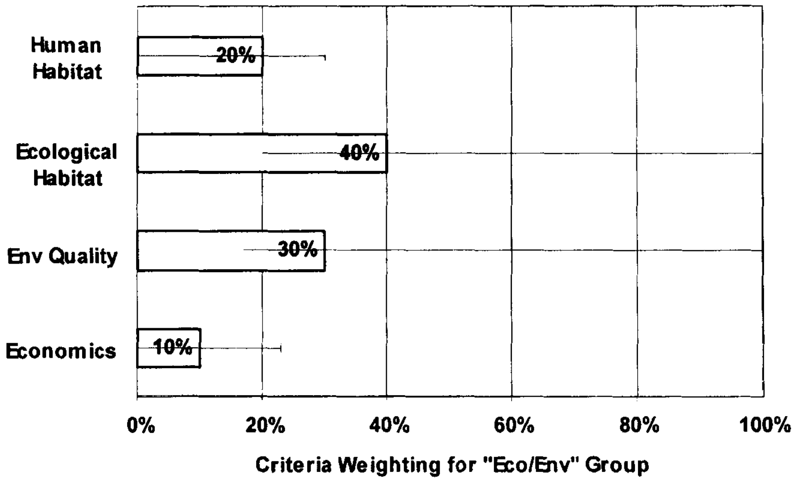


Figure 4. The alternative ordering suggested by the 'eco/env' group was extremely stable. This may reflect the fact that the views of this group are strongly held (and therefore easy to elicit and interpret), and that the alternatives available are easy to align with respect to their expressed priorities.

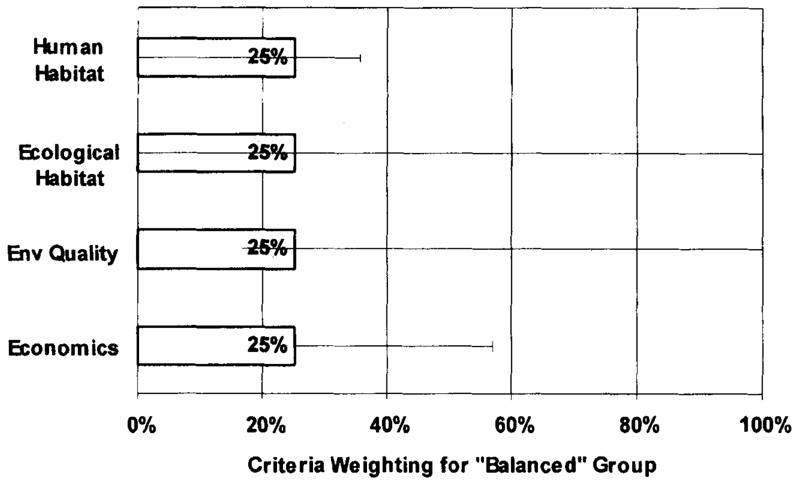


Figure 5. Only a slight underweighting of economic criteria would break the tie (which matched elicited responses) between wetlands restoration and the upland capped cell, suggesting that the 'balanced' group may be more willing to compromise on wetlands restoration than the capped cell as a valid second best alternative.

3) The **Commerce** (Figure 7) group was concerned with maximizing economic benefits after being assured that the alternative was environmentally sound, and was predicted to prefer:
   e. Cement Manufacture (+0.52, -0.13)
   f. Upland Capped Cell (+0.48, -0.35)
   g. Wetlands Restoration (+0.31, -0.40)
   h. Flowable Concrete Fill (+0.10, -0.54)
4) The **Human Health and Habitat** (Figure 9) saw human health and well-being as the most important consideration and as an indicator of environmental well-being. The positive and negative flows results did not correlate well for this group, suggesting that the cement manufacture and wetlands options are not comparable. The results depend upon whether positive or negative flows are used to create the ordering. They are listed here on the basis of net flows:
   a. Wetlands Restoration (+0.49, -0.20)
   b. Cement Manufacture (+0.30, -0.20)
   c. Upland Capped Cell (+0.38, -0.33)
   d. Flowable Concrete Fill (+0.03, -0.47)

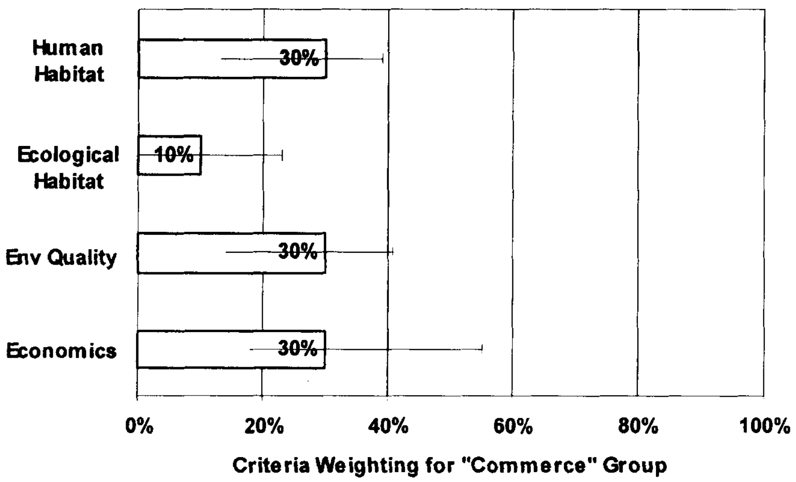


Figure 6. Like the 'eco/env' group, the 'commerce' group results are fairly stable over a wide range of criteria weightings. Severe reduction of the environmental quality criteria weighting would result in a complex ordering of results in which a decision (between cement manufacture and upland capped cell) depends entirely on the willingness to trade off cost versus human habitat.

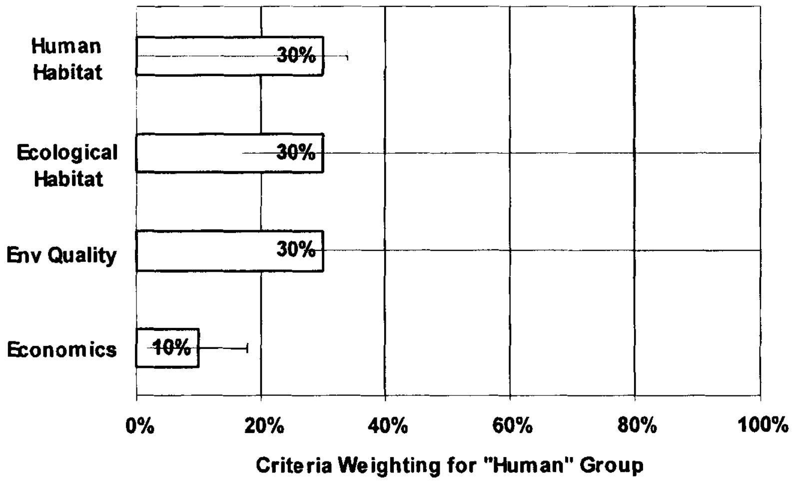


Figure 7. All criteria except environmental quality are fairly stable for the 'Human' group. Reduced weighting of env. qual. would move the upland capped cell into a dominant position, suggesting that this alternative may be perceived as a valid second-best compromise.

Stakeholders were not equally divided among these groups, but each group included more than one individual, and all individuals were readily placed in a group. Stability intervals were calculated as the range of percentage weightings for any single criterion over which the alternative orderings would remain unchanged. Figures 5-8 (below) show the criteria weightings representative of each group, the alternative order, and (as horizontal error bars) the range over which the order is stable.

To investigate the potential alliances and conflicts, the results can be presented in tabular form, or graphically by *Decision Lab 2000.* Figure 8 is a multidimensional depiction of the different alternatives and groups that places stakeholder groups along approximately independent axes. The alternatives are plotted (as triangles) in relation to these axes so that the group which most prefers an alternative will be pointing towards the triangular symbol representing that a alternative. None of the groups are pointing towards the flowable fill, which represents their uniform dissatisfaction with the performance of this alternative in this case study. The 'pi' axis represents the average of all stakeholder positions, but you can see that this axis is not pointing directly at any alternative, suggesting that a compromise solution may not be close at hand. As may not be surprising, the 'eco/env' group and 'commerce' group are nearly in opposition – presented at obtuse angles to one another. The 'eco/env' group will surely oppose the 'commerce' group's first choice: upland capped cell. However, the sensitivity analysis suggested that the 'human health' group may be willing to compromise on the upland capped cell, which was also flagged in a tie (with wetlands restoration) for second-best by the balanced group. In fact, this analysis suggests there may be potential support among three of the four stakeholder groups for the alternative actually planned for in the Cocheco project: the upland capped cell.

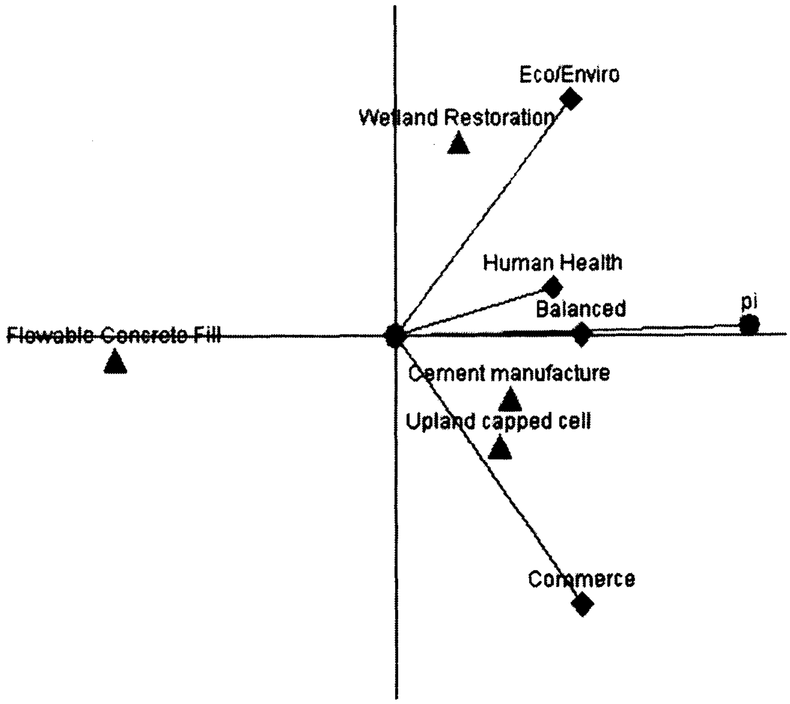


Figure 8. *Decision Lab* can portray the competing alternatives in a multidimensional graphic called a GAIA plane, which shows the potential conflicts between different stakeholder groups by aligning their axes in opposing directions.

4. Conclusions

The principal purpose of the MCDA approach employed is not necessarily to find the 'best' decision, but to improve the understanding of different stakeholder values. The approach of eliciting these values in parallel to development and assessment of the alternatives at hand is unusual, but may allow for smoother introduction of new technological alternatives (such as beneficial reuse of contaminated sediments) at a more fully developed point in the decision process. So long as expert assessments of the new technologies are consistent with the criteria and metrics established in conjunction with stakeholders, the outranking methods presented may provide an effective tool for assessment of which stakeholder groups may be likely to support the new alternative, or where potential compromises (or opposition) may be discovered. In progressing this research, the following general observations may be made:

1. The stakeholders involved were eager to have their values heard and incorporated into the management decision process, but critical of written survey methods (although they did confirm the effectiveness of the survey at conveying a simplified, basic message).

2. The research experts recognized the importance of stakeholder values to management of environmental problems, but were especially challenged by the process of devising measurable, quantitative metrics that would faithfully reflect the decision criteria expressed.
3. The systematic outranking analysis is more effective at identifying dominated alternatives (such as flowable fill in this case), discovering the sensitivity of second-best alternatives to preference weightings, and in general sorting out complex trade-offs than are stakeholder or expert heuristic processes.

The stakeholder value elicitation / public participation and decision analysis process studied may have potential for other environmental managers as a guideline on how to cost-effectively incorporate the public into the decision process in a meaningful way.

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ANALYSIS IN SUPPORT OF ENVIRONMENTAL DECISION-MAKING

Report of the Working Group on Environmental Decision-Making[1]

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Abstract

Comparative risk assessment (CRA) is an important environmental decision-making tool, often used to identify broad risk categories and high priority risks. This chapter addresses some of the challenges that analysts and risk practitioners face in using CRA to promote scientifically sound decisions about environmental priorities. We argue that in order to meaningfully inform the decision-making process, analysts must give serious consideration to the decision context, and they must carefully tailor their messages, with a view towards clarity and balance.

1. Introduction

Much of what falls under the collective heading of Environmental Decision-Making can be classified as efforts to *identify* and *quantify* environmental risks, and to set priorities and implement policies for *managing* these risks. During the course of the past decade, comparative risk assessment (CRA) has emerged as an important tool for identifying broad risk categories and risks of high priority. The application of CRA methods generally entails the use of science and professional judgment. Collectively, these methods and judgments are used to group risks into well-defined categories – from the most imperative risks to the least pressing. In this way, CRA provides a framework that facilitates the formulation and framing of risk management options.

Used appropriately, CRA can be a useful tool for promoting scientifically sound decisions about environmental priorities. Our remarks in this chapter are directed

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at practicing analysts who are interested in improving the relevance and impact of their risk assessment work to the policymaking process. There is much that risk analysts can learn about speaking scientific "truth" to political "power," starting with an appreciation that neither truth nor power are absolutes in the real world of environmental decision-making [1, 2]. The overarching message of this chapter is that risk analysts will successfully inform environmental decisions only if they pay close attention to their context, assumptions, and communication strategies. Indeed, effective analysis will often depend on *political* as well as *scientific* sophistication.

In most public policy settings, decision-making is a *deliberative* process. In this way, analysis in support of environmental decisions works best if it is tailored to the particular features of the policy setting and context. Typical models of decision processes distinguish among three distinct stages of analysis: *identification* of the need to make a decision, developing *alternative choices*, and *selecting* a preferred choice [3, 4]. During the identification stage, we distinguish between the task of recognizing that a problem exists, and diagnosing broadly that someone should act. During the development stage, decision-makers can search for existing, "on-the-shelf" options, or they can generate new policy options. During the selection stage, experts may perform analysis and evaluation, followed by a policy choice process that focuses on the judgment of a single decision-maker or the negotiated outcome of several decision-makers. In many instances, this sequence is followed by an authorization process that seeks to legitimize the decision. Interruptions may occur – and analysts can provide input – at any one of these stages. Typically, the analysis process starts with a scoping stage to define appropriate boundaries for analysis. Next comes the hard work of doing credible analysis and evaluation. Finally, analysts need to communicate their findings, mindful of the decision context in which they find themselves.

The case of global climate change provides a useful vehicle for illustrating these ideas. Governmental and intergovernmental decisions as to what actions, if any, should be taken in response to the prospect of anthropogenic climate change are characterized by conflicting criteria and large degrees of uncertainty. Policymakers must identify and evaluate a broad range of possible response options, in a decision context where the potential effects of climate change may be recognized only decades hence. In addition to being characterized by long lead-times, some of these effects may be irreversible. Also complicating efforts to arrive at a robust set of response options are the inherent nonlinearities that characterize the global climate system. The existence of such nonlinearities forces decision-makers to consider the possibility of "shocks" or "surprises" in the climate system, which could potentially give rise to catastrophic consequences.

As with any complex, real-world problem, there are many possible ways to frame the greenhouse problem. Typically, the framing of options focuses on identifying strategies for reducing net emissions of key greenhouse gases. Such reductions can be achieved by either reducing the *sources* of greenhouse gases or by increasing the *sinks* (natural or otherwise) of these gases. The formal evaluation of greenhouse gas abatement strategies then focuses on three main issues:

- The economic costs of pursuing specific greenhouse gas abatement strategies;

- The social benefits of abating or mitigating global climate change;
- Uncertainty concerning the level and timing of global climate change, the effectiveness of abatement strategies, and both opportunity costs and social benefits.

Situations like this, where an environmental policy choice must be made among alternative courses of action with uncertain consequences, are referred to as *decision problems under uncertainty.* Following Leonard Savage's classic formulation, a decision problem under uncertainty consists of four basic elements:

1. A set $A = \{a_1,\ldots,a_m\}$ of alternative *policy options,* one of which will be selected;
2. For each policy option $a_i \in A$, a set $U_j = \{X_1,\ldots,X_n\}$ of *uncertain events* that describe the possible outcomes associated with the selection of policy option a_i;
3. Corresponding to each set U_j is a set of *consequences* $C_j = \{c_1,\ldots,c_r\}$;
4. A *preference order* $\leq$, defined as a binary relation between some of the elements of A.

Having chosen a policy action $a_i \in A$, we observe the occurrence of uncertain events in the set U_j. Each uncertain event in U_j has associated with it a corresponding consequence set C_j. In this way, the set of uncertain events U_j forms a partition of the total set of possibilities, with each policy option a_i mapping elements of U_j to the elements $c_k \in C_j$. As alluded to earlier, scientific knowledge and professional judgment both play pivotal roles in defining the set A of possible policy options. In a similar vein, risk assessors focus much of their activity on characterizing and evaluating the sets U_j and C_j.

The remainder of this chapter explores different aspects of how practicing risk analysts can best match analysis to context. The next section takes a descriptive approach, focusing on some of the defining characteristics of real-world decision contexts. Section 3 focuses on normative considerations, exploring the topic of decision rules and their application to environmental decision-making problems. Section 4 takes a prescriptive approach, combining the lessons and insights derived from the descriptive and normative viewpoints. The chapter concludes in Section 5 with a brief commentary on the challenges that risk analysts face in supporting the environmental decision-making process.

2. Decision Contexts: A Descriptive View

Although there are an infinite number of dimensions for classifying decisions, a few suffice to illustrate the importance of considering the context of analytical decision support. Examine the dimensions shown in Table 1. Some are characteristics of the decision-making entity itself: Is it an individual or shared activity? Do the decisions take place within a hierarchal or egalitarian structure? Other dimensions are situational characteristics of the decision problem: Is it a stand-alone or sequential decision process? Are the time horizons long or short? Are the decisions routine or strategic, focused on one decision criterion or several? Finally, there are several dimensions that

have links to the decision situation itself: Are there spillovers effects? Are the impacts local, regional, or global? Are the potential consequences irreversible?

Consider, again, our global climate change example. In this instance, the decision-making unit could be an individual or a group. A lone farmer, for example, worried about the potentially adverse effects of climate change on his crops, might contemplate the purchase of crop protection insurance. On the opposite end of the spectrum, the decision-making unit could be an intergovernmental body, charged with making recommendations as to what actions should be taken by world governments to manage the potential threat of global warming. Looking at some of the other dimensions shown in Table 1, we note that the climate change problem is characterized by long time horizons, complex trade-offs, potentially global and irreversible impacts, and various levels of incertitude.

Moving, now, to a very different context, consider the decision to build a large hydroelectric facility, such as Egypt's Aswan Dam. With the benefit of hindsight, we can see that this was a shared decision, taking place within a hierarchal structure. To a certain extent, it was a stand-alone decision, with a very long time horizon, balancing multiple criteria. Impacts were large-scale, practically irreversible, and characterized by a high degree of uncertainty. It turns out that there were significant spillovers affecting various parties. The analytical requirements were immense, even more so than the dam's planners initially realized, especially concerning spillover effects such as schistosomiasis and downstream nutrient flows.

These examples suggest the ways in which the decision context, in effect, *sets* the analytical tasks: It frames the analysis and its scope, it indicates which types of tools are likely to be needed, and it guides the communication strategy that transmits the findings to the relevant decision-makers.

3. Decision Rules: A Normative View

In any environmental decision-making situation, there is – in addition to the descriptive context – a normative context for analysis. When analysts support decision-makers, their work should adopt assumptions and values that are acceptable to those decision-makers. As one of us has previously written:

> One way to characterize the division of labor between decision makers and analysts is that decision makers decide on reasonable decision rules, while analysts strive to apply those rules rationally. "Reasonable" decision rules are internally consistent and are the outcome of moral argumentation. "Rational" application is logical, valid, reliable, and empirically tested [5, p. 34].

Reasonable decision rules must be judged "reasonable" in context. What are some widely used decision rules, and how do we determine whether they are appropriate for a specific context? In considering this question, we return, once again, to our climate change example. A normative stance on this issue might, for example, hold that policymakers should endeavor to identify climate policies that *minimize expected social loss,* over all options $a \in A$ being considered as part of the deliberation process. With this as a decision criterion, let us assume that we are interested in only one uncertain

TABLE 1: Dimensions for Characterizing Environmental Decision Contexts.

Dimensions	Categories	Examples
Decision-making Unit	Individual, Group, Organization, Institution	Person, family, employer, national government
Decision-making Structure	Hierarchal, Egalitarian	Environmental control investment in a firm; Choice of fertilizers & pesticides in person's back yard
Formality	Informal, Formal	Adjusting the thermostat controlling comfort in a building; Specifying performance standards in a supply contract
Sequencing	One-Off, Sequential, Strategic	Siting a nuclear waste repository; Issuing a hazardous waste transport permit; Adopting the precautionary principle
Time Horizon	Short, Medium, Long	One month, five years, one generation
Tradeoffs	Single decision criterion, Multiple criteria with no conflicts, Multicriteria tradeoffs	Least-cost solution, Eco-efficient solution, Sustainable solution
Public-ness	Private, Private with Spillovers, Public	Purchasing book; Driving or selling a polluting car; Creating a national park
Extent of Impacts	Individual, Local, Regional/National, Global	Eating contaminated food; Noise pollution; Acid rain; Climate change
Reversibility	Reversible, Reversible in Long Run, Practically Irreversible	Over-fishing a river, Stratospheric ozone depletion, Allowing species extinctions
Incertitude	Stochastic Uncertainty, Structural Uncertainty, Scientific Ignorance	Confidence in estimating: person's risk of death by lightning; price of flood insurance in 10 years; impacts of climate change on Mediterranean region

quantity, X (say, e.g., the uncertainty concerning the level or magnitude of global climate change by 2050), and that this uncertain quantity is characterized by Pr(X), the probability mass function for X. If event x occurs, and if policy option a is adopted, then the resulting social loss is represented by the function $l(x, a)$. Let l^* denote the minimum expected social loss. When x is a discrete random variable, our normative decision rule for computing l^* is given by

$$l^* = \min_{a \in A} E[l(x, a)]$$

$$= \min_{a \in A} \sum_{x} l(x, a) \Pr(x).$$

In addition to this class of decision rules, three other normative considerations are worth discussing briefly: *Efficiency*, *Equity*, and *Accountability*.

Efficiency

Underlying most utilitarian and microeconomic policy arguments is the following notion of efficiency: Given available resources, how can we maximize net social benefits? Benefit-cost analysis and risk-benefit analysis typically adopt this decision rule. Typically bundled in with it are strong assumptions, including: (i) *additivity*, such that costs and benefits experienced by individuals can be legitimately aggregated to determine total social welfare; and (ii) *infinite substitutability*, such that individuals who lose out can always be acceptably compensated by those who gain as a result of a decision. There are variants on this definition of efficiency, such as Bentham's "greatest good for the greatest number," or Pareto's "no losers test," which adds a unanimity requirement (parties must voluntarily agree to the transaction) to the net social benefit test. Analysts need to evaluate whether net social benefit, the no losers test, or some other decision rule is most reasonable for the decision context they are operating in.

Equity

Fairness is also of great importance in public decisions, but it can be defined in many different ways. Kant recommends intrinsic respect for fellow humans: never use persons only as means to some other objective, but always as ends in themselves. Rawls' egalitarian conception of justice argues that we should favor the most vulnerable members of society, "for there but for the grace of God go I." Nozick advocates equality of opportunities, whereas Gray argues that equality of outcomes is more important. Of course, equitable processes do not guarantee equitable outcomes; for this reason, analysts need to understand how decision-makers evaluate equity or fairness in specific situations or contexts.

Accountability

Minimax regret is a decision rule that tries to minimize the difference between the best and worst outcomes in an uncertain world. Public officials sometimes fear the evaluation of history; they want to know that they did not make seriously suboptimal decisions. A minimax regret decision rule favors options that reduce the variance in potential outcomes, thereby managing risk and enhancing the appearance of accountability. Analysts need to determine whether potential variance in outcomes is of concern to decision-makers. For a concise, plain-English discussion of alternative decision rules, and for references to the authors mentioned above, see Ref. [6].

Most environmental decisions incorporate numerous value judgments, in addition to the decision rule itself. The same is true of analysis supporting environmental decisions. Analytical scope, temporal window, and the extent of impacts considered – as well as the decision-making perspective adopted and the degree of aggregation in the results – are all value judgments that analysts make, preferably in consultation with their intended audience. In short, analysts need to actively manage the normative content of their work, seeking to justify, wherever possible, the normative adequacy of their assumptions and models [5].

4. Advice: A Prescriptive View

Normative theories of choice provide guidance on how people *should* make decisions, if they wish to act in accordance with certain logical principles. Often, there is a discord between normative theories of choice and how people *actually* behave in real-world decision contexts. A well-known normative model, for example, is discounted utility theory, which assumes that people discount future outcomes at a constant rate. What this means is that outcomes occurring in the future are valued less than outcomes (of similar magnitude) occurring in the present. In consequence, each year is valued proportionally less than the previous one. What empirical research has shown, however, is that most people do not exhibit constant discount rates over time; rather, their discount rates *decrease* over time, and are more hyperbolic than exponential in character [7].

Prescriptive decision analysis seeks to guide decision-makers toward consistent, rational choices, all the while recognizing their cognitive limitations. In this way, prescriptive approaches to environmental decision-making utilize descriptive theories of how people actually make decisions in real-world situations to inform and guide the way that normative theories of choice are used in evaluating complex environmental decision problems. As Bell et al. [8] explain, prescriptive analysis tries to answer the question of "how can real people – as opposed to imaginary, idealized, super rational people without psyches – make better choices in a way that does not do violence to their deep cognitive concerns?"

Often, it is difficult for decision-makers to imagine the long-term impacts of their decisions. In the case of global climate change, for example, Arrow et al. [9] argue that – in addition to the problems that long time horizons pose – climate change related policy decisions are complex, in large measure because

- The economics and scientific uncertainties are great;
- Many of the potential effects are irreversible;
- The problem is global in scale;
- There is a time lag between actions and their effects.

Approaching these challenges from a prescriptive perspective, climate researchers and environmental policy analysts have sought to explore how the evaluation of climate change response options can be framed as *sequential* decision problems, where successive, interdependent greenhouse gas abatement decisions are made over a finite time horizon [10]. As we described in Section 2, many real-world decision problems are characterized by sequences of successive, interdependent decisions. Viewing the climate change problem from a sequential perspective requires that we first recognize that *an optimal course of action will depend upon the optimal policy choices made at subsequent decision points*. From a prescriptive vantage point, then, *near-term* action is not taken without first considering what climate change response options might be available in the future, as well as what might be observed in the long-term about important climate change-related uncertainties. One must recognize, also, that both mid- and long-term abatement actions will – in some measure – depend on the observed consequences of short-term policy actions.

As these remarks suggest, each environmental decision-making situation calls for a unique set of considerations with regard to the development of prescriptive frameworks for guiding policy choice. The development of such frameworks often requires a quixotic blend of perspectives and disciplines, with a "dual understanding" of both the substantive technical aspects of the problem and a nuanced understanding of the political dimensions of the problem. At a foundational level, prescriptive modeling requires that analysts strike a meaningful balance between scientific adequacy or realism, on the one hand, and model transparency, model complexity, and ease of communication, on the other.

5. Conclusions

The case studies presented elsewhere in this book demonstrate that CRA can usefully inform environmental decision-making, but that such an outcome is by no means guaranteed. To improve the chances of success, analysts need to tailor their methods and approaches to their specific context. That context has "factual" aspects (such as the scope and the nature of the decision) and "values" aspects (such as the appropriate decision rules and other framing assumptions). Both aspects need to be managed actively by analysts.

Finally, we end by returning to the call made at the start of the chapter for increased political sophistication among analysts. If analysts do not have access to decision-makers — if power is not listening to truth — then analytical efforts may not matter much. "The notion that the force of the better argument should prevail has particularly shallow roots in human experience" [5, p. 24]. Analysts need to be able to recognize when they are wasting their time, and find alternative ways to make their work matter.

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Appendix: Recommendations for Future Work and Workshops

The authors co-chaired several meetings of the Environmental Decision-Making Working Group. Working Group participants identified recommendations for future work at NATO workshops, as well as possible collaborative activities among the researchers attending the workshop. Key recommendations that came out of these discussions include the following:

- Perform a set of national CRAs nested within a Mediterranean CRA, with possible topics including: water quality (including links to public awareness), water re-use, water scarcity, solid waste landfill siting and remediation, agricultural practices, air pollution (and other industrial impacts), location-specific multimedia CRA, "Global CRA" comparing aggregate risks across countries, influence of non-force threats for national security (e.g., ecological security).
- Develop guidelines for CRA best practices; Document lessons from the past — learning from disasters — with a view towards preventing future problems.
- Document the practice of environmental impact assessment across countries.
- Test the intercultural relevance of CRA use and communication strategies.
- Develop CRA communication strategies for use in Mediterranean countries.
- Investigate sustainability of urban areas.
- Investigate water extremes (floods and droughts).
- Conflict resolution as a tool for managing trans-boundary water issues.

Part 3

Case Studies in Risk Assessment and Environmental Decision Making

WATER QUALITY CHALLENGES FACING EGYPT

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Abstract

Water in sufficient quantity and of adequate quality is necessary for the well-being of all living organisms. The importance and intensive use of fresh water makes it a vulnerable and increasingly limited resource. A wide range of human activities may lead to environmental deterioration of surface and ground water, either directly or indirectly.

In arid and semi arid regions, water resources management issue turns to be more imperative and necessary due to the scarcity of water resources and the irregularities of water flows in time and space. Deteriorating water quality is a particular threat in countries with scare water resources.

As far as Egypt is concerned, adequate supplies of fresh water is critical to the long term, sustained growth and development. Historically, water resources management focused on reallocating water to when and where it was required, a supply-side approach. In recent years, it has become increasingly apparent that the quality of available water is as important as the quantity. Poor water quality can render available supplies unsuitable for its intended uses. Thus, water quality, if not adequately managed, can serve as a serious limiting factor to the future economic development and to the public health and the environment which will result in enormous long term costs to the society. This in turn could lead to irreversible damage to the quantity and quality of available water resources. Thus, the need for better management of the quality of water resources is greatly recognized.

Recently, water quality issues have received high attention in the overall water resources planning and polices in Egypt. A water quality management program, with a national perspective was developed and implemented, which is based on an integrated approach to water quality data collection, analysis, interpretation, management and coordination. This program will provide a strong scientific basis for sound policy development and decision-making and assist in the development of strategies to reduce current and avoid future water quality problems.

In this context, the paper examines the main aspects and problems concerned with water quality deterioration in Egypt along with its environmental impacts and constraints to sustainable development. The importance and role of monitoring, data management, technology transfer, institutional strengthening, collaboration between ministries and stakeholders and sound financial framework are examined. The efforts exerted by the Ministry of Water Resources and Irrigation for better water quality

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Comparative Risk Assessment and Environmental Decision Making, 335–347.

management are highlighted. Finally, the challenges and benchmarks that decision makers have to face and deal with, for future actions are briefly outlined.

1. Introduction

Water is a daily touchstone in the life of every citizen, sustaining health, economic development and ecosystems. The Nile River represents a unique water system. It plays a key role in providing the main potential for all economic activities in the country. Pressure on the Nile water is already severe. The availability of water of acceptable quality is limited and getting even more restricted, while at the same time the need for more water as a result of population growth, industrial development and of cultivation of desert land is increasing. The intensive water use, disposal of untreated wastes and increasing demands for more water called for a new dimension for the water quality management consisting of a coherent set of measures of a different nature: technical, legal, institutional and financial.

Within the framework of a national strategy on water quality management, the Ministry of Water Resources and Irrigation (MWRI) is executing a water quality program with a national perspective to identify sources of water pollution, suitability of surface and groundwater quality for different uses, assessment of institutional, legal and financial frameworks and means of public participation. This is based on an integrated approach to water quality data collection, analysis, interpretation, management and coordination.

This paper addresses three main subjects, namely, the water quality problems and their impacts and constraints to sustainable development; the priorities and actions relevant to formulation of an action plan; and the benchmarks for future actions.

2. Major Challenges Confronting Water Quality Management

Egypt is confronting numerous constraints and challenges impeding the achievement of a sustainable water quality development and management. Among these challenges are:

- Population plays a fundamental role in questions about future water availability, use and quality. The population was doubled in the last 40 years from 33 million in 1965 to 67 million in 2001 and is expected to reach between 90 – 100 million in 2025 increasing the demand for scarce water and arable land;
- Most of the easily accessible water resources are already developed and exhausted, and what is left is the costly hardships;
- The complexity of the existing water problems and the fragility of resources, particularly groundwater aquifer;
- Diversity of pollution sources to surface and groundwater due to increased industrial and agricultural activities and insufficient sanitary facilities;

- Improper development projects in the past have produced devastating effects on the quality and sustainability of water bodies. Most of these projects were based on economic considerations with no regard to their negative and accumulative impacts on water bodies;
- Research results are often poorly accessible between ministries. This is partly attributable to the absence of good communications and to patent rights;
- Several technologies of wastewater treatment and industrial process water are inadequately introduced or adapted, particularly in respect to cost and energy reduction issues;
- Fragmentation of water related institutional infrastructure, overlapping and conflicting functions and competing interests of the concerned government authorities;
- Most of the prevailing legislation and regulations are not properly enforced;
- The water sector in respect to exploration, research, development, technologies and management requires considerable investment far larger than presently allocated. The most fundamental problem is the insufficiency of financial resources.

In the vague absence of comprehensive and realistic national water polices and long term plans, the water resources situation will remain a crucial issue.

3. Water Quality Problems

Water pollution is one of the most intractable problems faced by managers. The principal causes of water pollution and quality degradation are well known: untreated or inadequately treated domestic and industrial wastewater, improper use of fertilizers and pesticides, solid waste disposal and unplanned urban and rural development. Other activities that may add to the pollution of the water system include navigation and tourism especially due to the riverine fleet. As things stand now, it is evident that pollution from both point and diffuse sources are diversified and will continue to prevail at a rather higher rate both in quantity and quality, if not properly controlled.

In rural areas, almost 80% of the population is connected to sewers, while 30% of the provincial towns have piped sewage system. However, these networks usually work above design capacity and most of them require rehabilitation. In rural areas, only 5% of the population is connected to sewers, while the remaining wastewater are discharged into the nearest waterways. Also due to the high rates of urban growth which witness continuous migration from rural areas to urban centers, the waste management cannot be adequately controlled.

It is clear that growing industrialization without adequate means for treatment of solid and liquid disposals is taking place resulting in an increase in the volume of effluents and toxic wastes and in the variety of toxic contaminants discharged into the waterways. Effluents of industries such as pulp and paper, food processing, textile finishing and chemical synthesis typically generate heavy pollution loads.

The agricultural activities, which utilize increased levels of various fertilizers, pesticides and herbicides, affect the water system, soils and health of the workers, and

could have a direct toxic effect on food supplies. Also, the presence of pesticides in groundwater is of increasing concern.

The reverine fleet, which comprises of over 9000 units, contributes to river pollution by oil and grease, as well as domestic waste inputs. Oil and grease are often toxic to aquatic life and may exhibit the transfer of oxygen.

Illegal polluting practices are numerous and widespread, where there is little or no possibility of direct control. These sources range from intentional dumping of night-soil, garbage, washing of animals and domestic utensils to seepage from landfills, run-off from animal farms and accidental releases of chemicals.

The kind of water pollution caused by these activities can be grouped as follows:

- Oxygen demanding substances (BOD, COD, TOC) mainly from domestic and industrial activities;
- Heavy metals mainly from industrial activities;
- Bacterial pollution from domestic and industrial activities;
- Salination (TDS, Cl) from agriculture;
- Pesticides (DDT, y-HCH) from agriculture and some industrial activities;
- Nutrients (No_3, NH_4, PO_4) from domestic and agricultural activities;
- Hydrocarbons (oil & grease) mainly from industrial plants and navigation;
- Other organic micro-pollutants (PCB, PAK) from industrial plants.

These pollutants can have potentially damaging impact such as:

- Water related health problems;
- Accumulation of toxic pollutants in water, sediments and aquatic life;
- Changes in salinity with major consequences for the aquatic ecosystem;
- Eutrophication especially in lakes;
- Reduction of the natural purification potential of the water system;
- Limitation to non-conventional water resources usage practices.

4. Hazard Assessment

It is important to understand the risks that contaminated water poses to the public, agriculture and the environment. For example, the conditions of risk that might affect the health are the existence of polluted surface or groundwater and a direct or indirect contact mechanism between water and human. Steps can then be taken to manage the risks to reduce the consequences.

In 1995, the National Water Quality Conservation Unit (NWQCU) at the National Water Research Center (NWRC) performed a qualitative evaluation for seven water quality parameters for the different regions of Egypt in terms of their effect on public health, agriculture, and the environment[1]. Their study implicated scoring of anticipated risks by group of experts in the field of water quality management. The results of this qualitative analysis indicated that water quality effects increase from the south in Upper Egypt to the Northern Delta and Fyoum area. In the Delta area, the Western Delta, is most affected. The study showed that pathogens generally have the

highest impact for all regions, followed by trace metals. Other parameters vary by region, with nutrients adversely affecting water quality in the Delta and northern lakes, and salinity effects are greatest in Fayoum. Oxygen related substances pose the greatest hazard in the northern lakes and the western part of the Delta. The evaluation also indicated that the environment is most affected in all regions, followed by health, while agriculture is rated as least affected.

In 1999, a Task Force from the Ministry of Water Resources and Irrigation (MWRI) was appointed to draft the water quality priorities and strategies of the Ministry[2]. Their study identified priority areas where high pollution and high chances of contact exist. Indicators used to assess pollution conditions were coli bacteria for pathogens and heavy metals and nitrates for toxins. The approach was applied to the Nile River, groundwater and drainage system. The study showed that the priority areas where high pollution and high chances of contact exist can all related to the larger urban conglomerates of the county. The high population densities and industrial activities in combination with insufficient sewerage and treatment facilities cause a high pollution load on surface and groundwater to the extent that there is a health hazard. The study identified the priority areas in need for pollution control actions as shown in Figure 1. The study emphasized also the need for protective measures for pollution vulnerable areas like Lake Nasser.

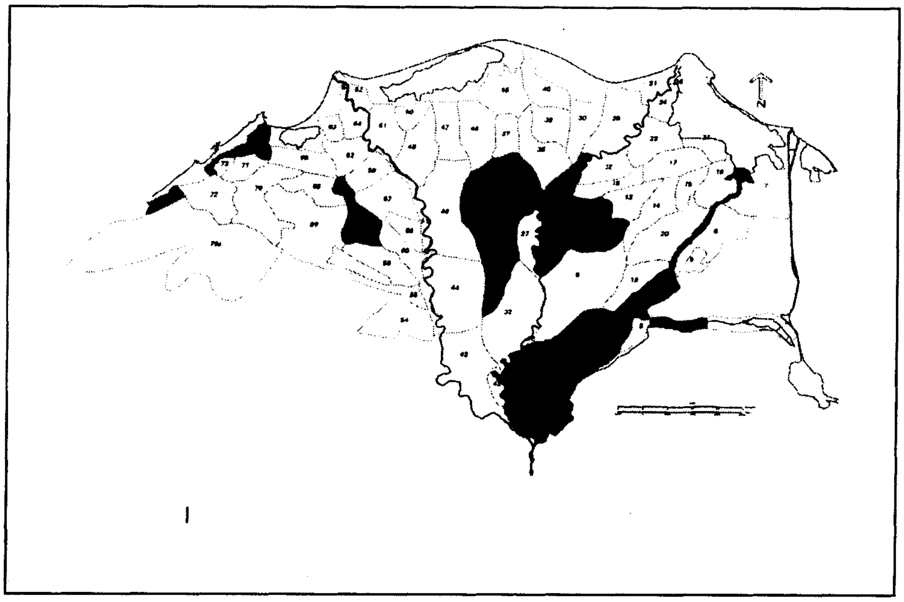

Fig. 1. Priority areas in the Nile Delta based on health criteria

5. Challenges and Actions in Water Quality Management

In view of the critical quantity and quality status of Egypt's water resources, appropriate solutions have to be taken to mitigate the prevailing problems and to prevent others from arising in the near future. The current and on-going debate on the water problems at the international, regional and national levels have revealed many recommendations and addressed adequate solutions to practically all water related problems. These recommendations which have been partially introduced in Egypt with varying degrees of implications are governed mainly by the prevailing economic and social conditions.

Thus, the Government of Egypt is adopting a clutch of measures for high priority problems that have an impact on human health or that lead to economic losses. Being the responsible authority in Egypt for the water resources planning and management, the Ministry of Water Resources and Irrigation is directing more efforts on integrated water management to improve the utilization of the water resources to meet the future increasing demand of acceptable quality. The Ministry identifies and delineates a water quality management strategy at the national level taking full and pragmatic accounts of the needs, priorities and constraints. This specific strategy reinforces the overall sustainable water management development.

5.1. WATER QUALITY MONITORING AND ANALYSIS

Monitoring networks and information system are important to underpin the formulation of policies, regulations, and environmental management. Well conceived monitoring can assure more efficient use of scarce resources through improved decision making. Effective control and reduction in pollutants requires an assessment of their loads and sources. Thus, information on concentrations and hence loads of the pollutants discharged into the waterways is essential.

During the past decades, three major water quality monitoring programs on the Nile River, drainage canals and groundwater were initiated by the research institutes of the National water Research Center (NWRC). These programs collected vast information on the quality characteristics of the concerned water bodies. In 1997, a need was identified to rationalize the water quality monitoring activities into a national monitoring program. Thus, in 1999, a water quality management program with a national perspective was developed and implemented for surface and groundwater, which is based on an integrated approach to water quality data collection, analysis, interpretation, management and coordination. This program is designed to contribute data, information and strengthen capacity. These elements provide the building blocks for the water resources planning and operation system in Egypt. The program provides a strong scientific basis for sound decision making and assists in the development of strategies to reduce current and avoid future water quality programs.

The national network has defined monitoring points, both stationary and mobile, and sampling frequency such as monthly for irrigation and drainage canals, seasonally for the Nile River or annually for the groundwater[3]. The range of parameters sampled for each of the water bodies and the required analytical techniques

have been identified. The network currently comprises of 245 locations for monitoring the surface water and 188 locations for groundwater as shown in Figure 2.

An important step taken by the MWRI towards the improvement of the reliability of the analyzed data is the construction and operation of the Central Laboratory for Environmental Quality Monitoring (CLEQM) within NWRC. CLEQM provides quality assurance and quality control analytical services for the national program.

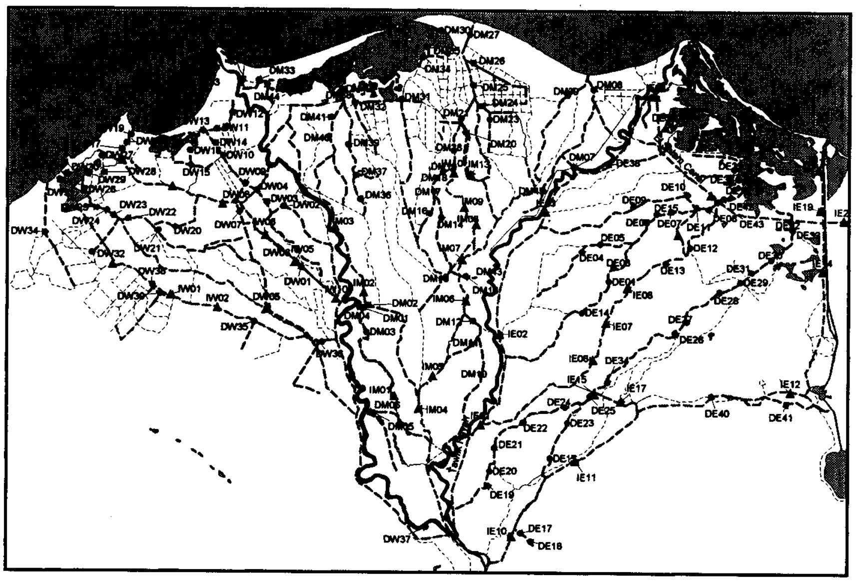

Fig. 2(a). Water Quality Monitoring sites (Delta)

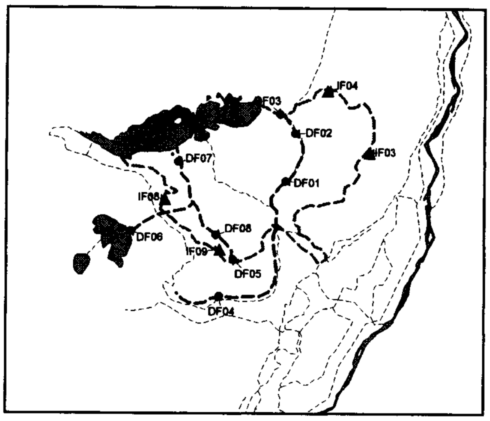

Fig. 2(b). Water Quality Monitoring sites (Fayoum)

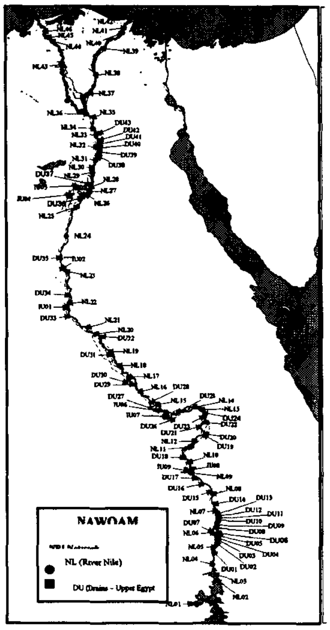

Fig. (c). Water Quality Monitoring sites (Upper Egypt and Lake Nasser)

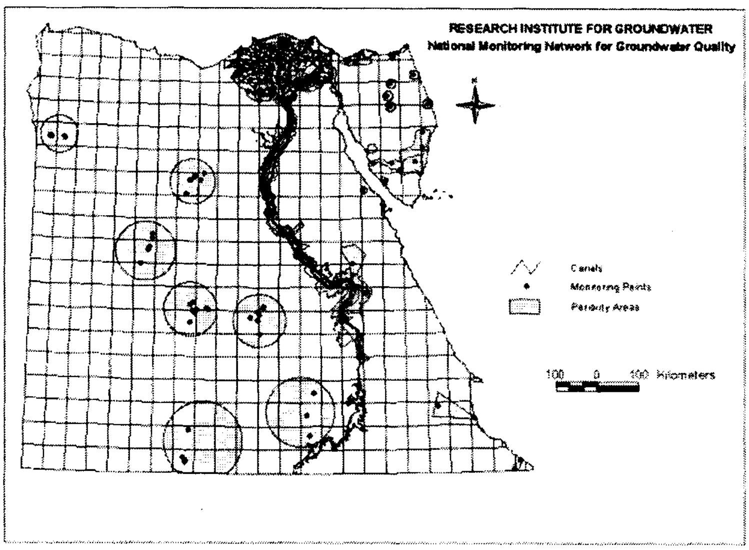

Fig. 2(d) Monitoring network for groundwater quality in priority and non- priority areas.

5.2. WATER QUALITY DATA MANAGEMENT

The development of an information management strategy and the implementation of a suitable database capacity is a key factor to the success of water quality management program. Major efforts are in progress to improve water quality data management so as to ensure widespread sharing of data and information, provision of meaningful data for decision making and standardization of data storage and reporting both within the MWRI and various ministries involved in water management. At present, water quality information is accessible through, for instance, yearbooks and status reports. A Fast Response Unit (FRU) is recently established at the NWRC to expedite the flow of information among and to the different institutions. A Central Water Quality Unit was established in 2002 at the MWRI that coordinates with the monitoring institutes and the law enforcement authority and is basically responsible for communications with other ministries. These actions will guarantee a free and fast exchange of information that is required for proper water management for operational and planning purposes.

5.3. GROUNDWATER MANAGEMENT

Groundwater is steadily gaining importance in the overall national water resources planning. Thus, the MWRI established in 2000 the Groundwater Sector which is responsible for groundwater licensing, development and management, and for preparation of a nation-wide plan for its rational management.

Work starts on the Groundwater Action Program which deals with both quantity and quality. One of the main themes is the integration of groundwater protection requirements into other policy areas, especially agricultural and regional planning.

5.4. NATIONAL WATER RESOURCES PLAN

In view of the critical quantity and quality status of Egypt's water resources, the MWRI is currently developing a National Water Resources Plan (NWRP) that describes how Egypt will safeguard its water resources in the future both with respect to quantity and quality, and how it will use these resources in the best way from a socio-economic and environmental point of view[4]. Alternative strategies are being developed and assessed with their costs and impact on socio-economic sectors and the environment.

5.5. INSTITUTIONAL DEVELOPMENT

The essential ingredient in capacity building is institutional development, including human resources. The capacity building process needs a strong commitment from all involved to support a comprehensive planning program. The MWRI has an adequate pool of scientists to support the basic water quality functions. Substantial short-term planning is provided to strengthen the skills of the existing staff.

6. National Protection Measures

Many initiatives have been taken by the Government of Egypt to protect the water resources and combat pollution. These actions are under implementation and (will) have positive impact on water resources and Egypt development.

6.1 AGRICULTURE SECTOR

- Removal of the Government subsidies on fertilizers and pesticides. This resulted in a considerable decline in the use of nitrogen and phosphate fertilizers.
- Promotion of pest control management. As a result, the overuse of herbicides to control aquatic weeds is now prohibited, and mechanical and biological maintenance are in practice.
- Development of an educational program to farmers on the proper use of pesticides and fertilizers, and information dissemination regarding the management practices that reduce the need for pesticides.

6.2. MUNICIPAL SECTOR

- Major programmes for sewage treatment plants for Cairo and Alexandria are on-going.
- An extensive program for the installation of new sewer systems and sewage treatment plants in smaller cities is being implemented. However, villages are hardly covered and poor functioning facilities are rarely renovated.

6.3 INDUSTRIAL SECTOR

- All new industrial communities are located in new cities in the desert areas and provided with sewage system. The industrial activities are based on new technologies developed for both source reduction and recycling.
- With respect to old industries, the government supports technology changes involving heavy capital investment.
- All industries discharging directly to the Nile were forced in December 1998 to take proper action to comply with Law 4 of 1994 as part of the plan sponsored by the Ministry of Environment to eliminate direct pollution to the Nile. However, more than 100 manufacturing establishments are still discharging polluted effluents to either agricultural drains which are mixed with freshwater for irrigation purposes or discharged to the Northern Lakes.

6.4. NAVIGATION SECTOR

- Provision of five on-shore facilities for collecting human wastes, bilge water and waste fuel from traffic and cruising boats at strategic locations along the river course.

7. Water Legislation

Effective programs to control water quality deterioration depends on the existence of adequate legislation supported by regulatory standards that specify the quality of water for the specific use.

The legal framework on water quality management in Egypt constitutes: (a) the recently enacted environmental law 4/1994 which addresses protection of water resources including control of pollution by harmful substances and related judicial procedures; (b) law 48/1982 which deals with the protection of the Nile and its waterways from pollution; (c) law 93/1962 which defines the standards for liquid waste disposal to sewers.

Enforcement of laws is affected by the competing interests of the concerned authorities whose mandate are intended for control of different end-uses and may not lay within the framework of water quality management. Governments often share similar enforcement responsibilities without clear delineation of authorities and powers. The overlapping of institutional function suggests a unified management and enforcement system.

8. Potential Measures for Solution

The main and foremost problem regarding water quality and pollution control is the absence of an integrated coordinated approach that is policy driven and takes into account agreed priorities. There is no joint strategy or action plan yet that coordinates the different tasks of the involved ministries.

The MWRI has made an important pace in this direction by formulating priorities, criteria and actions, involving other related ministries in the formulation of the short term action plans. It is hoped that coordinated strategies and long term action plans can then be formulated.

It is apparent that many challenges exist that decision makers in the water sector have to face and deal with for future actions. Among the potential measures for solution are:

- Bridging the present gap between water supply and sanitation coverage particularly in rural Egypt, and implementing new projects to cope with the anticipated increase in population. NOPWASD has a long term plan for constructing treatment plants and rehabilitating the old ones. A coordinated action plan with the MWRI is required. Investments are required to provide improvement in the most urgent areas.

- Closure of substance-cycles and changes in the use of raw materials and products can be stimulated by extra taxes on polluting products, incentives or tax cuts on clean products.
- The principles of "Polluter Pays" would provide the government with the funds to provide the necessary treatment.
- Many people are responding to real or perceived problems of water quality by consuming bottled water at cost that could be more usefully directed to water quality improvements.
- It is appropriate to have a broad policy with increasing protection to areas that are particularly vulnerable and which, at the same time, distinguishes between the more and less sensitive areas.
- Classification of water bodies according to their functions and uses and provide targets accordingly.
- Research in water treatment in the developed world has made considerable progress in recent years, both in the production of water of high quality and in wastewater treatment. A lack of research and little adaptation to significant advances in the developing countries have together resulted in almost stagnation in technology. Therefore, the research has to be local or at least adaptive of imported technology.
- Awareness raising of the value of water and hazard of pollution to all stakeholders is a prerequisite. Promote community involvement in water quality management to enable identifying actual needs, proposing response and fostering sense of belonging among the benefiting citizens. Awareness campaigns could be directed through the media, schools, training institutions, professional associations and political fora.
- Pollution reduction actions are the responsibility of more than one ministry. Overall coordination of country programmes is the primary task for success. Regular meetings between related parities should be held to share policy, practice and operational guidelines and to determine a coordinated approach. To ensure a good coordination the priorities should be agreed upon between these ministries and actions coordinated so that an optimal and efficient use of resources will lead to quick results. Also, continuous coordination between the different water sectors is essential to avoid duplications and ensure better quality products. Regular meetings between related parties should be held to share policy, practice and operational guidelines and to determine a coordinated approach. The presently developing cooperation between MWRI and Ministry of State for Environment is promising.
- The limited budgets available for water quality management require a careful analysis of the priorities that can and must be set, and a practical and pragmatical consideration in budget allocation is recommended.

9. Conclusion

Water is a finite and precious resource essential for sustaining life, for undertaking economically productive activities and for the environment. However, water is under potential severe threats from human, industrial and agricultural activities. The on-going decrease of water quality has severe implications, not only for water resources availability and human health, but also for vital ecosystems.

The scale and urgency of the challenges presented by deterioration of water quality in Egypt is clear. Many initiatives are being taken by Egypt aimed at radically improving the water quality, but these efforts need the informed support of all citizens. What is needed now is more timely and coordinated interventions by all actors and parties, from an enlighted and informed public and NGO's, to government decision makers in different ministries at various levels.

Participation, cooperation, commitment and consistent policy vision of all parties is a prerequisite for successful implementation of water quality management strategy. Clear strategies are the ideal framework for reaching complementarity which aims at a more sustainable use of limited resources by sharing lessons, avoiding unnecessary duplication and making use of a synergistic effect in different areas of expertise. Recently, mobilization of resources is witnessed in strengthening the institutional, legal, technical abilities to help the country in moving towards water quality management and pollution prevention of water bodies.

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RISK ASSESSMENT OF OCCUPATIONAL EXPOSURE TO PESTICIDES

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Abstract

The agricultural chemicals commonly labeled as pesticides are perhaps the largest group of poisonous substances being intentionally disseminated throughout the environment. For some pesticides neither health nor environmental risk evaluations are available. Therefore, at the moment the prevention of occupational and environmental consequences of pesticide use may only be achieved if methodologies and threshold environmental values are developed for the assessment of risk to the individual due to handling pesticides. Pre-marketing preventive actions are the primary responsibility of industry and the public health and governmental authorities. These include discovering the toxicological properties of each pesticide (hazard identification), determine the dose-response relationship (No Observed Effect Level, or NOEL, identification), assessing or predicting the exposure level in the various exposure scenarios; and characterizing the risk. Post-marketing preventive activities consist of the promotion of proper risk management at the workplace. Such management includes the safety assessment of the specific conditions of use, the adoption of proper work practices, and assessment of background exposure, cultural and life-style factors, and bio-markers of specific susceptibility. Such bio-markers including semen quality assessment and biochemical markers of exposure, serum uric acid, urea, creatinine, bilirubin, aspartate amino transferase (AST), and Glutamic Pyruvic Transaminase (ALT). In a case study (Kafr El-Sheikh Governorate, Egypt), involving 240 different individuals, reduction in semen quality in the pesticide applicators (PA) was seen compared with none farm workers (NFW). Also, biochemical markers, uric acid, urea, creatinine and AST in PA were near the upper limit values of normal. The monitoring and surveillance of pesticide exposures is mainly suggested by the established concept of the reference value and related analytical procedures. This concept is an essential contribution to an objective discussion of risk with regard to individual stress and strain profiles in environmental exposure scenarios.

1. Introduction

Pesticides are chemical substances designed to kill a variety of living organisms that humans consider undesirable. They are widely used throughout the world for protection of agricultural crops and in public health to control human diseases transmitted by

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vectors or intermediate hosts. Because of their high biological activity, and in some cases their persistence in the environment, the use of pesticides may cause undesired effects to human health and to the environment, especially in developing countries. The improper handling of some pesticides may result in severe acute intoxication; in some cases, adverse health effects may also result from long-term low-level exposures [1-2].

Inadequate control of pesticides had led to unacceptable situations such as the use of highly acute toxic pesticides without appropriate protection (Figure 1), pesticides of inferior quality, and deficient packaging and labeling. Pesticide users under such situations generally lack adequate knowledge of the safe and efficient use of pesticides [3].

Fig. 1. No training, no protection: workers in the developing world suffer the worst exposures with the least resources to prevent ill health.

Human data from occupational or accidental studies may be of value for the evaluation of pesticides. Persons involved in the manufacture or application of pesticides are in general more highly exposed than the general public and can therefore be monitored to identify toxic effects and their reversibilities, dose-response relationships and target organs and tissues [4].

In research, subjects with pathological enzyme often complained of various non-specific feelings of impaired well-being, including fatigue, exhaustion, lack of drive and concentration, sometimes itching and tingling skin, etc. Observations during the pesticide-handling period recorded the most striking results among the known symptom carriers and not among those individuals receiving specific exposure. Assessment of the test data in the light of their bio-markers confirmed that the half-lives of the active ingredient in the blood had a significant and often underestimated influence on the extent and severity of the feelings of impaired well-being. Early hints regarding metabolic modulation caused by enzyme deficits can usually be derived from the protein adducts induced by background exposures [5-8]. For this reason, direct or indirect measurement of the activities of essential enzymes involved in the metabolism of the foreign substance, together with measurements of the background exposure, are recommended as part both of primary and secondary prophylaxis of exposure. This is defined as before pesticide handling and especially in the early stages of handling [9].

2. Dose-effect relationship

Pertinent to any study of pesticide-related toxicity is the establishment of a dose-effect relationship for the agent, identifying the populations at risk and the possible range of concentrations to which they might be exposed. This is a basic tenet of the discipline of toxicology. Exposed individuals will include accidental and/or suicidal poisonings, agricultural workers (manufacturing, mixers/loaders, harvesters, handlers, etc.), bystanders inadvertently sprayed or exposed to off-target drift from spraying operations, and the general public [10]. With pesticides, it is extremely difficult to find a "clean study" with exposure to only one chemical, the usual situation being multi-chemical exposures. The single-chemical exposures tend to be individual case reports, poorly documented and signifying little unless one starts compiling them and a pattern appears that should not be ignored.

For individuals exposed to lower concentrations, the ranges become quite broad. It has been stated that, if no discernible adverse health effects are seen at high levels of exposure, it is unlikely that anything will be observed at lower levels. While this hypothesis may be valid for acute systemic effects, it is not applicable to chronic toxicity where latent changes in organ function, mutagenesis, carcinogenicity, or reproduction may occur at lower than those required to elicit acute toxicity.

3. Bio-markers of exposure to pesticides (Case study)

The field of pesticide toxicology is composed of a wide variety of chemical classes having diverse physicochemical properties as well as very different and, at times,

surprising mechanisms of action. The mechanisms by which some of these chemicals exert biological effects are known in great detail, but many others are still being studied in attempts to identify the biochemical or physiological event that initiates the toxic reaction. The differing biological or pathological effects of comparable pesticide exposure are seen as the consequence of the fact that some enzymes essential to metabolism are polymorphic. The concentrations of the exposure markers in the various matrices are appropriate for prophylaxis of any particular pesticide-handling process only in those cases where they can be oriented and assessed at the threshold doses of the unchanged active ingredients [9]. Biomarkers including semen quality assessment and biochemical markers of exposure, serum uric acid, urea, creatinine, albumin, bilirubin, aspartate amino transferase (AST), and alanine aminotransferase (ALT) were evaluated in 240 different individuals living in Kafr El-Sheikh Governorate, Egypt were evaluated in the present study.

3.1. MATERIALS AND METHODS

3.1.1. Study Participants

Participants in the present study worked and resided in Desuq City, Kafr El-Sheikh, Delta region, Egypt. All respondents were enlisted in the study during spring of 1998. These included 120 pesticide applicators (PA) and 120 non-farm workers (NFW), ages between 20-45 years. All participants were surveyed by written questionnaire regarding both health status and pesticide use. The health survey included family history, illnesses, hospitalizations, tests and medications, and a section consisting of 36 questions addressing organ system problems and life-style issues. A physician performed the clinical examination.

3.1.2. Blood Samples and Assays

During the spring and summer (the period during which farmers begin applying pesticides to their fields) of 1998, medical doctors collected their blood samples in Desuq City. Blood was drawn from the vein of each subject into 2 10-ml tubes (1 EDTA and 1 serum) that were held in ice water for shipment to the research laboratory. Samples were collected from each of the 120 participants. Serum was obtained by centrifugation at 3000 rpm for 20 minutes. Uric acid was determined using the method of Fossati et al. [11], urea according to Lazaroff [12], creatinine as per Hinegard and Tiderstrom [13], Bilirubin, Fossati et al., [14], and glutamic pyruvic transaminase (ALT) and glutamic oxaloacetic transaminase (AST) according to Bergmeyer [15].

3.1.3. Determination of semen parameters

Prior to submission of the specimen, married patients were given the instruction that abstinence of two days but not more than five days is recommended. Ideally, the total number of days of abstinence should reflect the person's usual pattern (i.e., if two days between intercourse is the usual pattern, then two days of abstinence should be allowed). The laboratory was notified of any medication the patient was taking. The specimen was collected in the laboratory, in a sterilized container. Once the specimen was collected, it was kept in an incubator. Semen viscosity, volume, viability, and fructose were measured according to Follas and Critser [16] and sperm concentration

and motility according to Yousef et al. [17]. Sperm morphology were measured according to Noe and Rock [18].

3.1.4 Data analysis

The differences between groups (PA and NFW) are compared using multiple t-tests and a chi-square test. Statistical computing was performed with the use of the SAS statistical package (SAS Version 6.12, SAS Institute, Inc., Cary, NC).

3.2. RESULTS AND DISCUSSION

The present study was carried out to determine the effects of exposure to pesticides on the hematological parameter sperm characteristics of pesticide applicators (PA) and 60 non-farm workers (NFW) in Desuq City, Egypt, in the Delta region. The results revealed moderate knowledge of the routes of absorption of pesticides and of potential symptoms following-exposure. Knowledge of personal protective measures was poor. Despite knowledge of some health risks associated with pesticides, the use of personal protective equipment was minimal due to financial constraints. Kidney (30%) and liver (38%) diseases were high among exposed workers (Table 1).

In the present study data revealed that a high number of pesticide applicators having uric acid (p <0.01), urea (p <0.05), and creatinine (p <0.01) values (>6mg/dl; >35 mg/dl and 1-1.2 mg/dl respectively) compared to non-exposed workers (Figure 2). These data are compatible with the percentage of kidney diseases in exposed workers (30%) compared to non-exposed workers (6.7%) (Table 1). More than one dialkylphosphorus metabolite was detected in almost all workers exposed to azinphos-methyl, and amount of metabolite found was correlated with high serum creatinine concentration of agricultural workers [19]. The positive correlation between exposure to the organophosphorus pesticides and creatinine level in previous studies agreed with that shown in the present study. In addition, serum creatinine and uric acid concentrations in exposed farm workers were significantly higher than the concentrations seen in the NFW (though the creatinine and uric acid concentration did not exceed the upper normal limit in all cases). The results of the examinations pointed to discrete lesions of the kidney [20]. Risk factors of pesticide poisoning hygiene and total absence of improper use of personal hygiene and total absence of improper use of personal devices were prevalent [21]. Insufficient protective clothing played a major part in increasing the number of incidence of kidney lesions in pesticide spray-workers in the present study.

The serum bilirubin concentrations (equal or more than 0.85 and less than 0.95 mg/dl) of pesticide applicators was higher (P <0.05) compared to non-exposed workers (Figure 3). On the other hand, there was no significant difference in serum ALT values between exposed and non-exposed workers (Figure 3). However, pesticide applicators had higher (P <0.05) serum AST activity, with a mean value 37 U/l or more than those of non-exposed workers (Figure 3). These results are in agreement with those shown in the previous study, which revealed a high percentage of liver dysfunction in workers exposed to pesticides (Table 1). The serum ALT and AST results are in accordance with that of Ballal et al. [22].

TABLE 1. Summary of questionnaire, including work hygiene, health history, and neurology history of pesticide applicators (PA) and control (NFW).

Questionnaire items			Control		Pesticide's applicators	
			No.	%	No.	%
Work hygiene	1	Smoking	32	53.3	44	73.3
	2	Carry cigarettes in work	—	——	40	66.7
	3	Smoking in working areas	——	——	35	58.3
	4	Carry food in working areas	——	——	56	93.3
	5	Eat or drink in working areas	——	——	56	93.3
	6	Wash hands by water before eating	——	——	52	86.7
	7	Wash hands by soap before eating	——	——	28	46.7
	8	Use protective devices	——	——	0	0.0
Health history	9	Back trouble	18	30.0	32	53.3
	10	Concussion	0	0.0	0	0.0
	11	Kidney disease	4	6.7	18	30.0
	12	Liver condition	15	25.0	23	38.3
	13	Arthritis or Rheumatism	4	6.7	15	25.0
	14	Skin disease	0	0.0	6	10.0
Neurology	15	Pain in neck	4	6.7	14	23.3
	16	Blurred vision	3	5.0	14	23.3
	17	Partial loss of sight in eyes	0	0.0	9	15.0
	18	Difficulty swallowing	2	3.3	4	6.7
	19	Sick in stomach	7	11.7	14	23.3
	20	Frequent vomiting	0	0.0	2	3.3
	21	Muscle weakness	8	13.3	12	20.0
	22	Frequent headaches	24	40.0	34	56.7
	23	Lost balance	0	0.0	3	5.0
	24	Dizziness	12	20.0	18	30.0
	25	Lost consciousness	2	3.3	8	13.3
	26	Difficulty to sleep	6	10.0	13	21.7
	27	Frequently felt tired	17	28.3	32	53.3
	28	Unexplained sweating	5	8.3	8	13.3
	29	Trouble in coordination	0	0.0	1	1.7
	30	Numbness in hand	1	1.7	3	5.0
	31	Numbness in feet	3	5.0	8	13.3
	32	Weight loss	2	3.3	4	6.7
	33	Repeating diarrhea	3	5. 0	12	20.0
	34	Tension	22	36.7	31	51.7

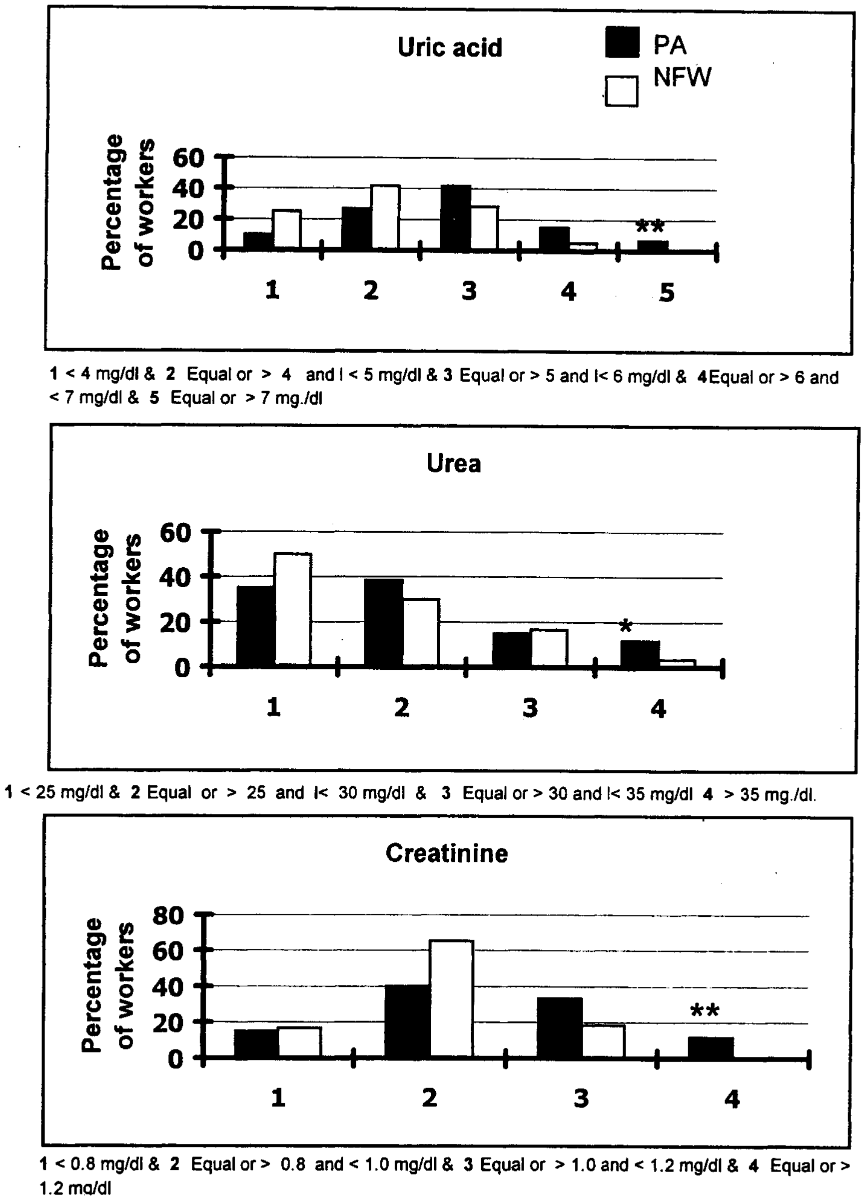
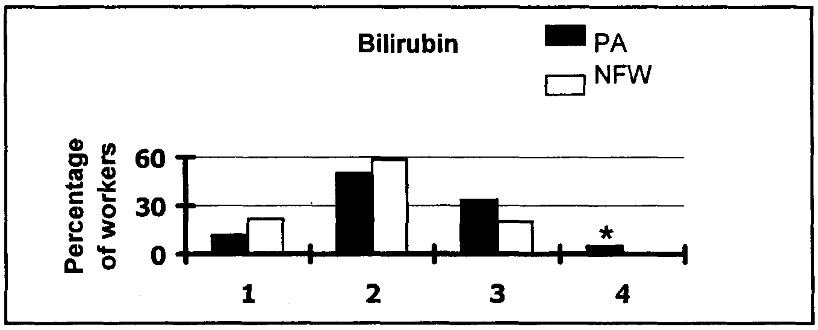
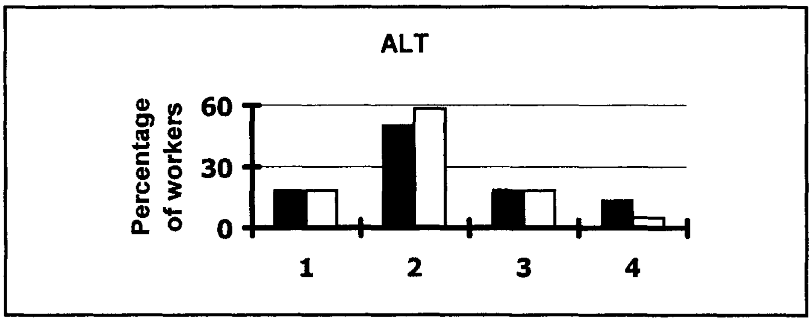


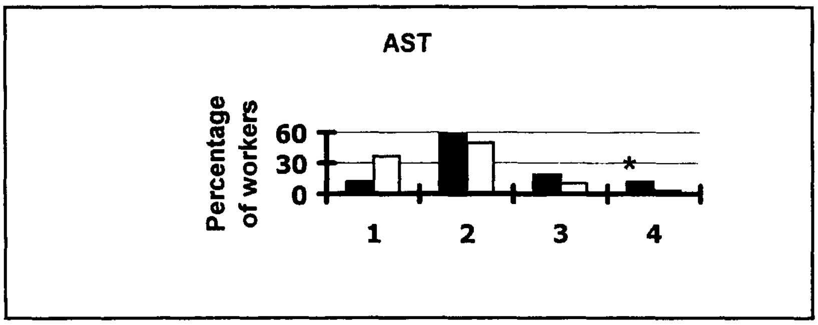
*Fig. 2. Percentage of male agricultural pesticides applicators (PA) and non-farm workers (NFW) of serum uric acid, urea, and creatinine values. **P <0.01; *P <0.05 versus matched NFW.*



1 < 0.75 mg/dl & 2 Equal or > 0.75 and < 0.85 mg/dl & 3 Equal or > 0.85 and < 0.95 mg/dl & 4 Equal or > 0.95 mg/dl



1 < 20 U/l & 2 Equal or > 20 and < 28 U/l & 3 Equal or > 28 and < 35 U/l & 4 Equal or > 35 U/l



1 < 23 U/l & 2 Equal or > 23 and < 30 U/l & 3 Equal or > 30 and < 37 U/l & 4 Equal or > 37 U/l

*Fig. 3. Percentage of male agricultural pesticides applicators (PA) and non farm workers (NFW) of serum total bilirubin, serum aspartate aminotransferase (AST), and alanine aminotransferase (ALT) values. *P <0.05 versus matched NFW.*

Many studies with other species have been carried out. Enan et al. [23] found a significant decrease in the serum ALT activity of rabbits given sub-lethal doses of profenphos. However, sub-lethal doses of cyanofenphos and profenphos and an acute single dose administration of profenphos resulted in an apparent increase in serum AST activity of rabbits. It was also observed that hepatic ALT activity was significantly increased in rabbits dosed acutely and subchronically with these insecticides. Also, Enan et al. [24] recorded a significant increase in serum ALT of rats after the administration of profenphos, parathion-methyl, sulprofos, malathion, dichlorvos and dimethoate. Significant inhibition in serum ALT of rats after the administration of leptophos, chlorpyrifos and diazinon was reported [24].

The disruption of transaminases from their normal values denotes biochemical impairment of tissue and cellular functions as they are involved in detoxification processes, metabolism, and biosynthesis of energetic macromolecules of different essential functions [25]. El-Gendy et al. [26] found that pyrophos and glyphosate caused spontaneous activation of liver AST and muscle ALT and caused significant inhibition of brain AST in common carp (*Cyprinus carpio*). Habiba and Ismail [27] reported that brain and muscle ASTs were inhibited in the New Zealand white rabbit fed on clover contaminated with profenfos, whereas liver AST was stimulated. Transaminases are important and critical enzymes in the biological processes. They play a role in amino acid catabolism and biosynthesis. ALT transfers the amino group of alanine to α-ketoglutaric acid, forming glutamic and pyruvic acids. Consequently, it is considered as a specific indicator of liver damage [23]. The possible mechanism involved in the elevation of ALT may be due to tissue damage, or due to increased synthesis or decreased catabolism of ALT [23].

The overall mean ejaculate volume and the sperm count of the NFW were higher ($P<0.001$) than PA workers (Table 2). The mean sperm count in NFW individual was 68.1×10^6, compared to 63.0×10^6 for PA (Table 2). These results agree with previous studies showing that semen quality to be reduced in men occupationally exposed to various pesticides [28-29]. To be normally fertile, an adult man needs to produce 100 million or more sperm every day. Any decrease in this output or in the functional competence of the sperm will lead to impairment of fertility potential. The concentrations of the sperm count in the present study did not reach this level. This may be due to indirect effects of many chemicals used in industry, agriculture, medicine and the home, which can potentially impair the process of spermatogenesis [30].

TABLE 2. The overall mean of sperm volume (ml), number (million/ml), abnormal shape (%), and semen fructose (mg/dl.) of agricultural pesticide applicators (PA) and non-farm workers (NFW). Values are mean ± S.E.M. **$p<0.001$ vs. NFW.

	Volume (ml)	No. of sperm (million/ml)	Abnormal (%)	Semen fructose (mg/dl)
PA	2.675 ± 0.09**	63.00 ± 5.22	18.35 ± 1.168**	353.90 ± 14.6
NFW	3.300 ± 0.15	68.10 ± 4.37	11.70 ± 0.678	362.30 ± 10.8

In the present study, PA sperm motility was highly affected when compared with NFW (Table 3). [Grade (0-1), which refers to low sperm activity, PA individuals, having high ($P<0.001$) sperm motility compared to NFW. Grade (3-4) showed a significant decrease ($P<0.05$) in PA compared to NFW (Table 3). Viability after one, two and three hours was monitored for both PA and NFW (Table 4). Sperm viability was lower ($p<0.001$) in PA compared to NFW. This reduction was significant ($p<0.006$ and 0.002) after 2hr and 3hr respectively as well. These results are consistent with those previously described that show fertility is reduced in men occupationally exposed to various pesticides [28, 31].

TABLE 3. The overall mean of sperm motility grade (%) of agricultural pesticide applicators (PA) and non-farmer workers (NFW). Values are mean ± S.E.M. **p<0.001 vs. NFW; *p<0.05vs NEW.

Grade	0-1	1-2	2-3	3-4
PA	13.25 ± 3.3**	18.75 ±3.4	32.50±3.4	36.75 ±5.36*
NFW	3.00 ± 1.8	12.75±1.4	28.50±2.4	52.50 ±4.59

TABLE 4. The overall mean of sperm viability (%) of agricultural pesticide applicators (PA) and no-farmer workers (NFW). Values are mean ± S.E.M. ***p<0.001 vs. NFW; **p<0.002 vs. NFW; *p<0.006 vs. NFW.

Viability	1hr	2hr	3hr
PA	56.10± 3.05***	47.10 ±2.98*	37.6 ±2.92**
NFW	66.45± 1.39	55.95±2.05	48.20±1.61

The deleterious effect of exposure to pesticides on sperm quality in the present study was also manifested by pronounced effect on sperm morphology (Figure 4 and Table 2). The most common types of sperm abnormalities noted in semen samples from PA were: double head, double tail, round head, droplet cytoplasm, and large head (Figure 4). The overall numbers of abnormal sperms in NFW were lower ($P<0.001$) than PA individuals (Table 2). Also, more than 50% of the PA had a high pus content compared to only 15% for NFW individuals. The high percentage of pus in NFW might be due to other sources of effects. It is generally agreed that large reductions in sperm number or large increases in sperm with abnormal shapes are associated with reduced fertility. Although this method of assessment is not as sensitive to small changes in sperm morphology, it is a reliable indicator of male reproductive toxicity [29]. The reason for sperm having abnormal shapes is not clear. Perhaps they are the results of a naturally occurring level of mistakes in the differentiation process or they may be the consequence of an abnormal chromosome complement [32]. The increase in the percentage of abnormal sperm observed indicates the genotoxic potency of pesticides exposure [29]. Others have shown a significant increase in chromosome aberrations in

sprayers when compared to unexposed persons [33-34]. These studies may explain the high number of abnormal sperms in the PA group compared to the NFW group.

Risk factors of pesticide poisoning, such as workers' ignorance regarding pesticide toxicity, poor personal hygiene, and total absence or improper use of personal protective devices were prevalent [21]. In fact, in the present study, workers have so far paid little attention to the proper use of pesticides. Insufficient protective clothing is contributing to pesticide poisoning among spray workers. Also, the majority of farmers and equipment workers never received any formal training prior to their first contact with pesticides and application equipment. Pre-employment and periodic medical examinations are recommended in addition to the formulation of health programs for all workers concerned.

Pesticide applicators are at risk of poisoning if there is no adherence to precautionary measures. The degree of human exposure depends on many factors, such as the type and toxicity of the pesticide used, the availability and use of protective clothing, and the duration of exposure [22, 35]. A very significant finding in the present study was the lack of specific protective clothing, which is essential for the safety of the exposed employee (Table 1). Human data from occupational studies may be of value for the evaluation of pesticides. Persons involved in application of pesticides are in general more prone to be exposed than the general public and can therefore be considered as monitors to identify toxic effect [36]. Pre-employment and periodic medical examinations are recommended in addition to the formulation of health programs for all concerned.

4. Conclusions

Bio-markers are a good tool for rapid risk evaluation of human pesticide exposure, especially in developing countries.

There is a need for better evaluation of health and environmental hazards and risks, and for training in risk management and risk reduction strategies, adapted to local conditions. Some national authorities recognize such shortcomings and are interested in acquiring the necessary additional expertise and infrastructure [3].

Although overall pesticide use may continue to grow, this will likely occur at a lower rate compared with the past. New products require lower application rates and have less environmental persistence and lower mammalian toxicity levels.There is a need for better evaluation of health and environmental hazards and risk, and for training in risk management and risk reduction strategies, adapted to local conditions.

A guiding principle is that the use of pesticides must be controlled in the public interest, and that one of the goals of regulating pesticides is to assure the availability of quality pest control products and their safe and efficient use. The key to the future appears to lie in the integration of pesticides into sound pest management practices, including proper selection of products (favoring any acceptable non-chemical alternatives). Also, adequate training of users, intensification of development of environmentally safer compounds, sound storage, transport and handling practices and, above all, use under concepts of IPM and "safe and efficient use" under Good Agricultural Practice (GAP).

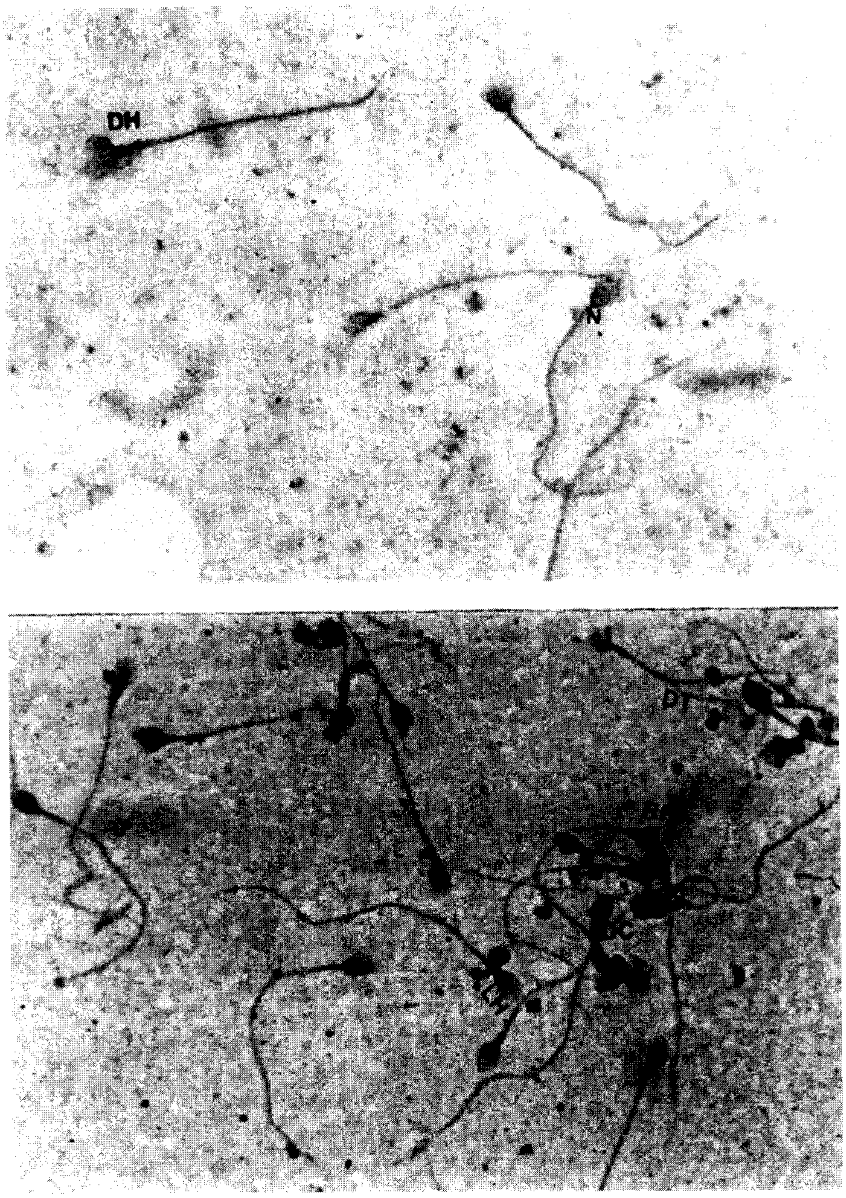


Fig. 4. Morphological features of spermatozoa in a semen sample of an agricultural pesticides applicator.. DH, duple head; N, normal; DT, double tail; RH, round head; DC, droplet cytoplasm; LH, large head.

Finally, we may conclude that, although progress has been made, a great deal still remains to be done to help those countries that require technical assistance to ensure, safety and effectiveness in pesticide use. If current trends persist, countries will have greater ability to use pesticides more efficiently and safely, as a component of last resort within the concept of IPM, in support of environmentally sustainable agricultural production systems, and in pursuit of an overall improvement of quality of life.

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6. References

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THE ROLE OF AIR POLLUTANTS AND SEWAGE WASTE IN ACCELERATION OF DEGRADATION OF THE ISLAMIC CULTURAL HERITAGE OF CAIRO

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Abstract

The impact of gaseous CO2 and SO2 attains additional importance when examining the processes of alteration and decay in historical buildings made of calcareous stones. We selected the Al-Ghuri complex (Cairo, Egypt) for the study of black crust formation on stone building surfaces due to the presence of carbonaceous materials and sulfatation via CaCO3 conversion to gypsum (CaSO4.2H2O). Under dry deposition, the breakdown of calcareous stones depends on the relative humidity and the presence of catalysts, e.g., O3 and oxides of Cu, Fe and Mn.

Depth profile analyses indicated the presence of sulfates to depths of 100 to 1200µm below the stone surface. Sulfatation was usually formed on the stone surface, and sometimes penetrated the outer surface through micro-fractures and pores. NaCl crystals (in cubic form) were precipitated at stone surfaces and pores due to the presence of sewage water via capillary action. Polyhalite $[K_2Ca_2Mg(SO_4)4.2H_2O]$ and epsomite ($MgSO_4.7H_2O$) were developed as minute crystals up to 12µm.

This study reveals the influence of sulfatation and salt formation and microrganismal activities on accelerating the rate of monumental stone degradation. Such processes involve reaction of stone with the pollutants, transformation of $CaCO_3$ to $CaSO_4.2H_2O$, and precipitation of NaCl, which under appropriate conditions crystallizes within pores and micro-cavities of stones. One possible consequence is the rupture and spilling of the upper surface of the stone due to crystal pressure.

1. Introduction

There are two motivations for research on the impact of polluted air on stonework: (i) preservation of cultural resources and national treasures, and (ii) accounting for material losses in terms of national economics. The first concern tends to emphasize

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stone buildings and monuments and their aesthetic values, whereas the second deals with common construction materials and the values of changes in their service lives.

There have been many papers on the mechanisms of calcareous stone and mortar damage (for example, Amoroso and Fassina, 1983; Livingston and Baer, 1983; Camuffo et al., 1983; Mossotti et al., 1987, Lipfert, 1989; El-Metwally et al., 2000). Lipfert (1989) noted that information on environmental deterioration of stone comes from at least three separate sources: environmental tests on stone samples, time-lapse measurements, and photographic comparisons of monuments and geological erosion data. In addition, data on the physical properties of relevant materials are available from laboratory experiments. Data from all of these sources indicate that calcareous stones have the highest rates of erosion and deterioration of the common building stones.

Lipfert (1989) also reported three mechanisms affecting the damage of calcareous mortar: (i) calcite dissolution in "clean" rain (pH = 5.6), (ii) dissolution due to acidic precipitation, and (iii) loss by conversion to soluble salts as a result of dry deposition of SO_2 and NO_x or other acidic species. Generally, rain is needed to remove the gypsum that has been formed.

Many authors agree that microbial colonization on stone will depend on essential factors such as: (i) the mineral composition of the stone that will provide the amount of extractable minerals available for the microbial growth, and (ii) the porosity and hygroscopicity of the rock. These factors can increase water uptake and facilitate the establishment of microbial organisms having high activity. The impact of pollution can decrease the colonization on stone, but can also increase the microbial growth of sulfate-reducing bacteria. According to Valentine (1993), biological agents, including algae, cyanobacteria, lichens, mosses, bacteria, fungi and high plants, constitute a complex community of organisms involved in the weathering processes of monumental stones.

Old Cairo City has been constructed on hills and low-lying areas frequently affected by flooding sewage water, which then degrades the Islamic Historical buildings. Also, they have been flooded by lakes and canal water; e.g., the former Egyptian canal (now Port Said Street). These water have had varying effects on the foundations and walls of the Islamic monuments.

The present study aims at studying the role of sewage water and polluted air on the degradation and decay of the Al-Ghuri Complex (1243-1249 p.m.) (Fig. 1). We also highlight here the risk assessment of man-made pollution on the Islamic cultural heritage. Our objective encompasses the characterization of microorganisms in weathered stone and in black crust on the monument surfaces. The Al-Ghuri complex is located at Al-Azhar and Al-Muizz Streets. The complex constitutes of the madrasa-mosque (called the hanging mosque). It is built in a cruciform plan and also contains four liwans, of which the qibla liwan is the largest. Sections of the mausoleum dome, kuttab, sabil, maq'ad, houses and wikala have been destroyed

2. Sampling and Procedures

Samples were collected from the walls at different heights from 30 to 75 cm above the floor from Samples were carefully packed and isolated from any environmental impact, *e.g.*, damp, high temperatures, atmospheric polluting gasses, and mechanical degradation. To help assess the effect of salt, the degree of degradation, and the thickness of alteration zone, and to estimate the degree and depth of the alteration of building stone, depth profile samples were prepared. In this process, four slabs are cut from the surface towards the sample interior, under dry conditions. On the basis of depth profiles, the investigation of salt crystallization and the role of SO_2 and CO_2 penetration of the stone up to few millimeters can be studied.

A binocular microscope was used to study the surface and profile samples. Scanning electron microscopy (SEM) was used to investigate carbonaceous materials phase transformation, salt formation, and microorganism colonization on the surface slab and depth profile samples. All slabs were analyzed with X-ray diffraction (XRD) to identify mineral phases. The LH Leybold CSA-5003 and LH Leybold CWA-5003 apparati [define what type of equipment this is] were used to analyze for sulfur, organic carbon, and CO_2 contents. The XRF technique was applied to determine the concentration of Fe_2O_3, Ni, Cu, Zn, Sr, Zr, Pb and Br in the depth profile samples.

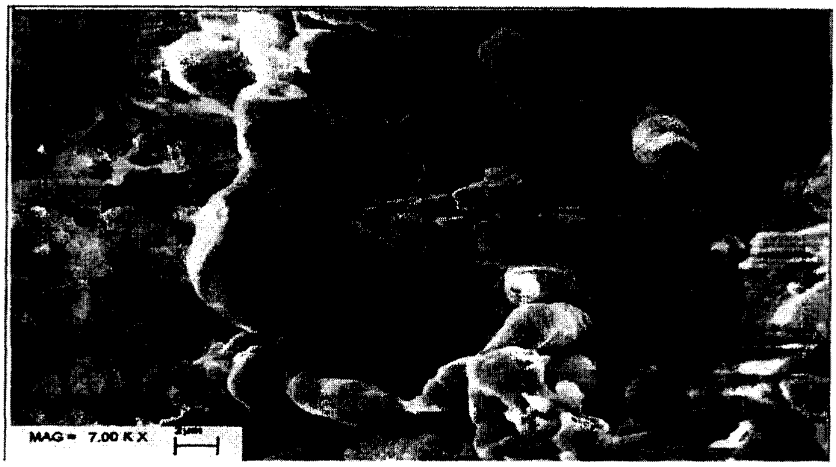
3. Results and Discussion

Macroscopic examination using the binocular microscope, petrographical analyses, and scanning electron microscopic investigation revealed mineralogical changes near to the stone surface. These included macroscopic and microscopic fractures and the lining or filling of pores to depths up to several hundred microns; the precipitation of halite in pores and micro-fractures; the occurrence of gypsum, epsomite ($MgSiO_4.7H_20$) and polyhalite ($K_2Ca_2Mg(SO_4).2H_2O$); and colonization by microorganisms. It is evident from Table 1 that gypsum has been developed up to depths of 1200μm. It may reach 12.2% of the stone surface (up to 8 mm) in some samples.

TABLE. 1 Results of XRD investigations of depth profile samples from the Al-Ghuri Complex.

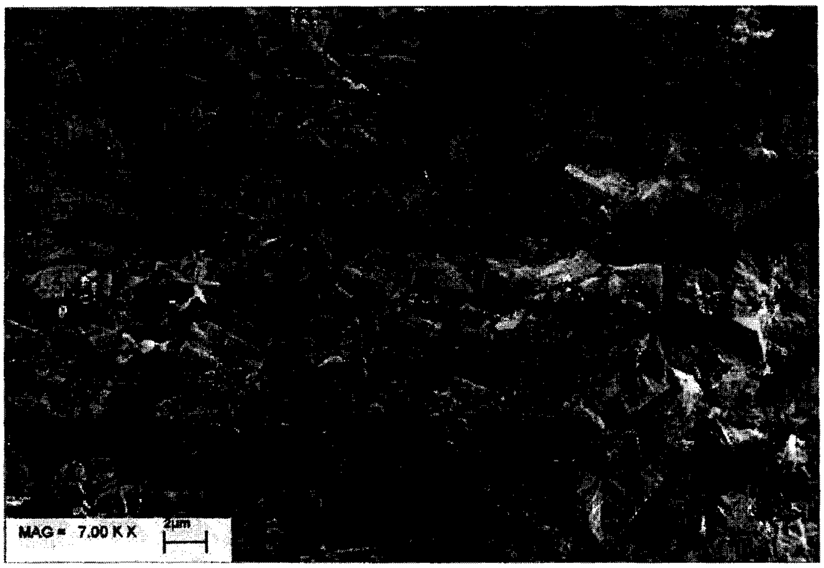
Sample No.	Calcite	Quartz	Gypsum	dolomite	Salts
$K_{1.1}$	47.1	4	2.9	42.2	3.8
$K_{1.2}$	49.3	6	1.7	39.8	3.2
$K_{1.3}$	54.1	3	0.9	40.2	1.8
$K_{1.4}$	46	10	0.2	43.8	--
$K_{1a.1}$	82.5	3.1	3.6	7.1	3.7
$K_{1a.2}$	88.9	2	2.5	4.2	2.4
$K_{1a.3}$	92.7	2	0.5	4	0.8
$K_{1a.4}$	90.5	3	0.5	5.8	0.2
$K_{4.1}$	91.7	2	1.7	--	4.6
$K_{4.2}$	94.8	2	0.5	--	2.7
$K_{4.3}$	97.2	1	0.6	--	1.2
$K_{4.4}$	98.5	1	--	--	0.5
$K_{4a.1}$	75.0	3	12.2	5.5	4.3
$K_{4a.2}$	89.7	5.6	2	--	2.7
$K_{4a.3}$	95.1	1	1	2	0.9
$K_{4a.4}$	96	2.5	0.5	1	--
$K_{2.1}$	81.7	2	1	12.3	4.1
$K_{2.2}$	72.8	5	0.7	18.2	2.8
$K_{2.3}$	75.5	3.2	--	19.7	1.6
$K_{2.4}$	72.3	6	--	20.3	1.4

The deterioration of building stones is mediated by pollutants derived from urban and human activities, but microorganisms and biological activities also play a role in the whole process. A considerable variety of organisms have been found colonizing stones (Fig. 2). It is evident that pollution inhibits microorganism (e.g. lichen) growth (Salvadory et al., 1991). These microorganisms produce a significant amount of oxalic acid through their metabolisms process and corrode stone by removing its calcium.



The precipitation and growth of halite (NaCl) on stone surfaces (Fig.3) and in pore systems (Fig. 4) up to 500µm, due to the seepage of underground water, is widely observed our study of the Al-Ghuri complex. XRD investigation of our samples indicates that their salt concentrations reach up to 4.3% weight units at the most outer crust of the stone surface, and decrease deeper in the rock (up to 1.2%). These values reflect the role of evaporation on the percent of crystalline salts. NaCl precipitation causes: (i) considerable increase of porosity of the building stone; (ii) increase of cavity and pore sizes; (iii) production and acceleration of microfracture growth in the surface of the lime mortars and stones due to crystal pressure; and (iv) acceleration of the rate of stone exfoliation.

The chemical analyses of depth profile samples from Al-Ghuri complex (Table 2) reveal interesting points. Transition and heavy metals (Fe, Pb, Zn, Cu, Ni and Br) generally increase in the outermost surfaces of the black crusts relative to the sample interiors. This is attributed to polluted gases derived mainly from diesel engine exhaust, which is primarily composed of soot and metallic particles bearing Fe and Fe-S as major elements and Cr, Cu and Ni as trace elements. These elements play a major role in the catalytic oxidation rates of SO_2 (Leysen et al, 1989; Rodrigues-Navarro and Sebastian, 1996; Ghedini et al., 200 and Chebas et al., 2000).


MAG = 7.00 K X
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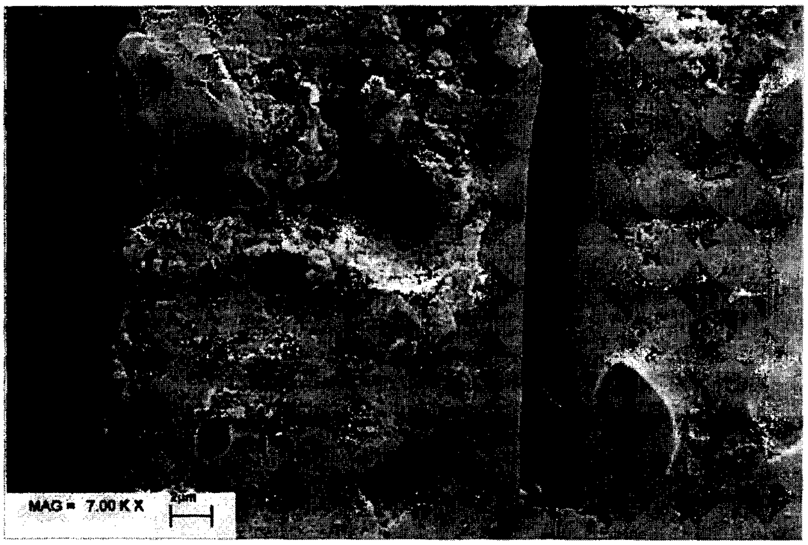
MAG = 7.00 K X
2µm

TABLE 2. Results of chemical analyses of depth profile samples from monumental calcareous stones in the Al-Ghuri complex.

Samp. No.	CaO wt%	Fe_2O_3 wt%	Organic carbon	CO_2 Wt%	S wt%	Ni ppm	Cu Ppm	Zn ppm	Sr ppm	Zr Ppm	Pb ppm	Br Ppm
K1-1	43.12	0.43	0.518	42.606	0.900	26	36	40	452	42	16	9
K1-2	43.79	0.37	0.415	42.938	0.099	23	30	34	712	36	13	7
K1-3	44.33	0.36	0.346	43.264	0.066	23	24	31	789	40	9	6
K1-4	45.01	0.27	0.215	43.711	0.054	21	20	31	818	39	8	6
K1a-1	42.63	0.25	0.218	42.270	2.033	29	31	37	637	39	19	11
K1a-2	44.21	0.46	0.119	45.009	0.401	26	25	32	918	36	15	8
K1a-3	44.64	0.43	0.090	46.127	0.000	25	21	30	1012	36	12	7
K1a-4	44.98	0.38	0.024	46.321	0.208	23	19	27	1105	42	12	7
K2-1	43.65	0.47	0.823	43.097	1.220	24	16	34	850	30	20	8
K2-2	45.20	0.39	0.384	44.327	0.053	21	14	28	916	32	15	6
K2-3	45.67	0.32	0.292	44.441	0.039	19	13	24	1216	33	11	6
K2-4	45.81	0.34	0.128	44.678	0.039	18	13	21	1679	29	12	5

The systematic depletion of CaO and CO_2 as a percent of weight from the sample interior towards the black crust is associated with sulfur enrichment, which causes the transformation of $CaCO_3$ to either precipitated or leached $CaSO_4.2H_2O$ and $Ca(HCO_3)_2$ (El-Metwally et al., 2000) The abrupt increase in organic carbon in the black crust reveals the deposition of carbonaceous material on the stone surface derived from fuel combustion and the metabolism of microorganisms, which colonize stone under damp and salty conditions. Pb and Br are usually derived from gasoline engines, which are widely used in Cairo. These two elements cause a lower rate of fixation of SO_2 as gypsum in limestone (Rodrigues-Navarro and Sebastian, 1996).

4. Conclusion

1 - Black sulfated crusts deposited on the surfaces of historic buildings and monuments are composed of inorganic materials (mainly gypsum) and a complex mixture of organic compounds. Generally the black crust coating the surfaces of building materials located in urban (polluted) environments is made up of matters present in aerosols and particulates, which are derived by dry and/or wet deposition processes. The organic compounds are trapped in the mineral matrix, where the activity of micro-organisms is also high.

2 - Wet $CaCO_3$ surfaces are good absorbers of gaseous SO_2, but deposition rates will diminish as surfaces become saturated with gypsum. Rain washing can remove gypsum and renew these surfaces because of the increased solubility of gypsum relative to calcite.

3 - Since the microorganisms thrive in a salty environment (Schostak et al., 1992), salt extraction, combined with the prevention of further salt input, will decrease microbial activity and thus bio-deterioration. Also, climate can be controlled to a lower humidity that will at least reduce the growth of microbes to an acceptable level. Lack of cleaning and maintenance of historic buildings and monuments is one of the major

factors contributing to biological colonization and deterioration. However, climate control to reduce microbial attack cannot be achieved in most cases, and therefore cleaning procedures and biocide application must be carried out.

4 - The pollution impacts lead to monumental stone damage due to surface loss and channeling by run-off, gypsum crystallization and calcite dissolution, and redistribution in pores and voids, which causes surface weakening.

5 - Sewage water plays a major role in the formation of halite either on the stone surface or in its pore system. The pressure of salt crystallization enhances the growth and development of micro- and macro-fractures and consequently leads to exfoliation of the stone surface.

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IRRIGATION WITH TREATED WASTEWATER IN ISRAEL -ASSESSMENT OF ENVIRONMENTAL ASPECTS

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Abstract

Population growth increases the output of sewage, which must be treated and discarded. Agricultural reuse can function as wastewater disposal, and simultaneously convert an environmental threat to a benefit, i.e., the supply of irrigation water to agriculture. Both quantity and quality aspects should be considered when irrigating with treated effluents, since its constituents may affect crops and groundwater. In this paper we refer to nutrients, represented by nitrogen or nitrates (NO_3), and salinity, represented by EC (electrical conductivity) or chlorides (Cl). The levels of these constituents in wastewater are usually elevated; therefore, they may affect crops, soil structure, or groundwater quality. Combining wastewater treatment and desalination processes to maintain groundwater quality can diminish these impacts and prevent environmental deterioration. We have assessed the environmental impacts of wastewater irrigation by focusing on nitrate and chloride constituents. To that end, we developed an economic-hydrological model that incorporates various water sources and treatment processes in order to analyze the accelerated contamination of groundwater.

1. Background

Population growth decreases the amount of available fresh water and increases the quantity of urban wastes that need to be treated and discarded. One optional solution is to reuse this urban wastewater for crop irrigation, thus providing an abundant and cheap source of water for agriculture.

Effluents contain many pollutants, including macro- and micro-organic matter, macro- and micro-inorganic matter, salinity, and pathogens (Wallach, 1994). Effluents reused in crop irrigation serve as a source of both water and nutrients, but they should be treated, and used cautiously to avoid potential damage to crops, soils, and groundwater. Conventional treatment processes at the secondary or tertiary levels can take care of most polluting constituents, including nitrates. However, salinity can be decreased only through special, relatively expensive desalination processes. These processes include reverse osmosis and electro-dialysis, which reduce both chlorides and nitrates.

Our analysis will relate to the Israeli experience with widespread agricultural reuse of wastewater, necessitated by the limited sources of fresh water. The water

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sources available for agriculture will decrease further with population growth, while wastewater sources will continue to increase. Treated effluents already serve as a very important water source in Israeli agriculture (Haruvy, 1997a, 1997b). Since urban wastes need to be treated and discarded anyway, agricultural reuse of effluents may serve not only to provide a source of agricultural water and nutrients, but also as means of preserving environmental quality (Haruvy, 1998). In the present paper we will focus on the salinity/chlorides and nitrates/nitrogen constituents as they affect groundwater.

2. Methodology

We have developed a model that assesses various aspects of the impact of wastewater irrigation on crops and groundwater. Salinity is quantified as the level of chlorides. We designed an approach to the economic evaluation of groundwater pollution; it is based on a hydrological model that predicts the flow of chlorides through the unsaturated zone of the subsoil and into the groundwater below. We assumed a threshold for chloride concentration in drinking water; when this threshold is reached, desalination of groundwater is applied. Desalinated water is mixed with other domestic water sources until the chloride concentration has been reduced to the permitted level. The resulting damage to groundwater by wastewater irrigation is computed as the additional costs of water supply, including water production, wastewater treatment and the preceding desalination.

This model was applied to an agricultural and an urban area. Several water supply alternatives were compared, including agricultural irrigation with wastewater combined with local aquifer water, imported aquifer water, and 'National Carrier' water. We computed the time variations of the resulting chloride levels, and the water supply costs for the various scenarios.

Water sources to town and agriculture include local groundwater, imported groundwater, the National Carrier, wastewater, seawater and rain.. Restrictions include a balance of leaching to groundwater with groundwater pumping. Salinity is computed as a weighted average of the contributions to the total salinity from all water resources. The costs are the sum of the costs of urban and agricultural water supplies; and the derived cost levels are compared for the relevant water supply alternatives.

We also applied this model to analyze the impacts of nutrients on groundwater. This impact is represented by the concentration of nitrates (NO^{-3}) in the groundwater (as mg/l).

3 Effect of chlorides on groundwater

Irrigation with effluents may accelerate the contamination of groundwater. We designed an approach to the economic evaluation of the impact of the concentrations of pollutants (as chlorides and nitrates) on groundwater by predicting the flow of chlorides through the unsaturated zone of the subsoil and into the groundwater below. The time needed for the completion of the chloride flow through the unsaturated zone is about 5 years close to the seashore of Israel, about 20 years in the central part of the Coastal

Plain, and ranges from tens up to hundreds of years in the southeastern part of the Coastal Plain.

We assumed that the threshold for chloride concentration in the water supply for human consumption is 250 mg/l (the current limits for the concentrations of chlorides (as Cl) in drinking water are 250 mg/l in Israel and 100 mg/l in Europe). We also assumed that desalination of groundwater by means of reverse osmosis technology is initiated when the concentration of chlorides in groundwater reaches the permitted threshold concentration (250 mg/l). Part of the groundwater is then desalinated to a Cl level of 150 mg/l, and is mixed with other domestic water sources until the threshold level is reached.

Economic assessment of the damage caused to groundwater by irrigation with effluents, as compared with the conditions of irrigation without effluents, is affected by the point in time at which desalination is applied.

This model was applied to a given hydrological cell in central Israel that is assumed to include an agricultural area of 1,211 ha of citrus crops and an urban area of 1,052 ha with a population of 120,000 inhabitants. The computed annual water consumption for agriculture is 9.1 MCM (7,500 CM/ha) and for the urban area is 12.0 MCM (100 CM per capita). Water leaching from the urban area amounts to 1.14 MCM; rain from the urban area to 2.78 MCM; leaching from the agricultural area to 1.82 MCM; and rain from the agricultural area to 2.66 MCM. Total recharge of the aquifer is 7.18 MCM/year, and this is also the amount that can be drawn from the local aquifer. The initial chloride concentrations (as chlorine) are 241, 350 and 10 mg/l in the groundwater, wastewater, and rainwater respectively. The town uses local aquifer water, whereas agriculture uses treated effluents (Table 1, scenario 1.1).

In scenario 1.1, which is based on wastewater irrigation, desalination costs are much higher because of the need to begin desalination processes earlier than in the other scenarios. However, since the cost of wastewater is lower, the total cost of water supply for the whole region seems relatively low, but the higher chloride levels in the groundwater in this scenario should also be taken into account (Table 2).

TABLE 1: Water Quantity and Chloride (Cl) Concentration of Various Scenarios

Source	Scenario 1.1		Scenario 1.2		Scenario 1.3	
	Quantity MCM	Cl Mg/l	Quantity MCM	Cl mg/l	Quantity MCM	Cl mg/l
Aquifer	488	241	488	241	488	241
Rain (mm)	550	10	550	10	550	10
Wastewater	9.08	350	0.00	350	0.00	350
National carrier	0.00	220	6.95	220	9.95	220
Imported aquifer	4.82	250	6.95	176	3.95	150

This basic scenario 1.1 is compared with scenario 1.2, in which the town consumes local aquifer water, imported aquifer water with a salinity level (as Cl) of 176 mg/l, and 'National Carrier' water with a salinity level of 220 mg/l. In this scenario,

agriculture uses aquifer surplus as well as the other two water sources. In the third scenario (scenario 1.3) the town consumes local aquifer water and national carrier water. The three scenarios are summarized in Table 1.

Resulting chloride levels are higher in the scenario based on wastewater irrigation (scenario 1.1) than in the other scenarios (Table 2). In the first basic scenario, 1.1, chloride levels in town water for domestic use are higher until the 40^{th} year; chloride levels in the aquifer increase after the initial stage, and the difference in chloride levels between scenario 1.1 and the other scenarios increases gradually with time. We calculated the water supply costs for the various scenarios under the assumption that desalination of the water supplied to the town is initiated at a threshold level of 250 mg/l (Table 3).

TABLE 2: Chloride levels through time in various water-source scenarios (mg/l Cl)

Year (from steady state)	Scenario 1.1		Scenario 1.2		Scenario 1.3	
	Aquifer	Town	Aquifer	Town	Aquifer	Town
1	241	245	241	220	241	231
10	275	250	251	224	252	236
20	307	250	261	229	264	242
30	335	250	270	234	276	248
40	359	250	278	250	289	250

TABLE 3: Calculated costs for various scenarios (present value in $ millions)

	Scenario 1.1	Scenario 1.2	Scenario 1.3
Total desalination cost	102.40	1.66	0.73
Discounted desalination cost	18.43	0.16	0.07
Supply cost to town	489.6	471.6	517.6
Supply cost to agriculture	295.1	381.1	381.1
Total supply cost	784.8	852.7	898.7
Discounted supply cost	273.7	310.3	327.7

4. Effect of nitrates on groundwater

The model described above was also applied to the case of pollution with nitrates, which is also accelerated by wastewater irrigation. In the Coastal Plain of Israel, nitrate leaching fractions are 20-45% and 20-60% from cropped land and orange groves respectively (Haruvy et. al, 1997), and about twice that for wastewater (Hadas et. al, 2000).

The annual damage caused by drinking water with a nitrate level exceeding 10 mg/l was estimated by Ready and Henken (1999) at about \$635 per family per year or approximately \$1.50/CM. Since treatment costs are much lower, this is a high estimate for a preventable damage. Yadav and Wall (1998) analyzed the costs and benefits of nitrate control in groundwater. Another approach was used to estimate the damage to groundwater; this approach was based on the decrease in agricultural profits caused by nitrate leaching restrictions (Haruvy et. al, 2000). The described model was applied to assess environmental damage caused by nitrate pollution, on the basis of the additional treatment costs needed for drinking water supplied to a town.

For the above-described basic scenario (scenario 2.1) we refer to the same basic regional characteristics as described earlier. Each year the city uses 7.18 MCM from the local aquifer and an additional 4.82 MCM is imported from another groundwater source, while treated effluents supply the whole agricultural water consumption of 9.08 MCM.

We assume that the nitrate concentrations of the various water sources in the initial scenario are as follows: local aquifer- 63 mg/l, rain- 2 mg/l, treated effluents- 143 mg/l, national carrier- 30 mg/l (diluted with some local water sources), and other groundwater sources (average)- 58 mg/l. We assume an increase by 1 mg/l annually, and also that it takes 14 years for the leachate to pass the unsaturated zone (Yaron et. al, 1999).

Typically, fertilization supplies 200 kg nitrogen per ha for all water sources, with 40% leaching of nitrate, and 100 kg of nitrogen per ha to the wastewater-irrigated area, with 60% leaching. According to Hadas et al. (2000), leaching from citrus groves is 20-60%, and the leaching rate from wastewater irrigation is twice that from fresh water irrigation, because of the increased organic matter. Hence, we have the following nitrate concentrations in leaching water: local aquifer- 362 mg/l, treated effluents- 606 mg/l, national carrier- 296 mg/l, and other groundwater source- 352 mg/l. The resulting amounts of leached nitrates are 100.7 metric tons from urban use and 1,114.7 metric tons from agriculture. The drawn nitrate amounts to 452.0 metric tons, and the nitrate added to the aquifer to 763.3 metric tons, i.e., 1.56 mg/l in the first year (after the leachate has passed the unsaturated zone).

The basic scenario (scenario 2.1) was compared with the following scenarios (Table 6):

In Scenario 2.2, the town still consumes 12.0 MCM but receives only half of it from the local groundwater; it also uses imported groundwater (25%) with a nitrate concentration of 58 mg/l (increasing each year by 1 mg/l) and obtains the rest from the National Carrier, assuming an overall nitrate concentration of 30 mg/l (the basic nitrate concentration in the National Carrier is 1 mg/l). For agriculture to obtain its needed 9.08 MCM, it takes the local groundwater that remains from the town consumption (an amount of 1.18 MCM) and satisfies the remaining demand equally from imported groundwater and the National Carrier (3.95 MCM each). In Scenario 2.3, the city gets its water supply equally from the local aquifer and imported groundwater. In Scenario 2.4, the city gets its water supply equally from the local aquifer and the National Carrier. In Scenario 2.5, the city water sources are the same as in Scenario 1, while agriculture does not use wastewater but obtains water equally from the National Carrier

and other groundwater sources. A comparison of the water sources for various scenarios is presented in Table 4.

TABLE 4: Water balance in the various scenarios (MCM)

Water source	Scenario 2.1	Scenario 2.2	Scenario 2.3	Scenario 2.4	Scenario 2.5
TOWN					
Local groundwater	7.18	6.00	6.00	6.00	7.18
Imported groundwater	4.82	3.00	6.00	0.00	4.82
National Carrier water	0.00	3.00	0.00	6.00	0.00
Total	12.00	12.00	12.00	12.00	12.00
AGRICULTURE					
Wastewater	9.08	0.00	0.00	0.00	0.00
Local groundwater	0.00	1.18	1.18	1.18	0.00
Imported groundwater	0.00	3.95	3.95	3.95	4.54
National Carrier water	0.00	3.95	3.95	3.95	4.54
Total	9.08	9.08	9.08	9.08	9.08
Grand total	21.08	21.08	21.08	21.08	21.08

Table 5 presents the nitrates concentrations in the aquifer as they change in the course of time. In scenario 1.1, based on wastewater irrigation, the nitrate concentration rises to 76 mg/l in the 10th year, 89 mg/l in the 20th year, 100 mg/l in the 30th year, and 110 mg/l in the 40th year. The nitrate concentrations are: higher in Scenario 2.3 than in Scenario 2.2, higher in Scenario 2.2 than in Scenario 2.4,, and higher in Scenario 2.1 than in Scenario 2.5. In Scenarios 2.2 and 2.5, about 237-259 metric tons of nitrate are added to the aquifer during the 1st year (after steady state), and the initial increase of nitrate concentration is 0.49-0.53 mg/l as compared with Scenario 2.1 with wastewater irrigation (adding initially 1.56 mg/l nitrate to the aquifer).

We also assessed the costs of supplying water to the region throughout a period of 50 years (beginning 13 years before the achievement of a steady state). For this, we assumed that the drinking water nitrate level restriction for the city is 70 mg/l. When this level is reached, the water is treated to reduce the level to 50 mg/l, and is then mixed with existing water sources to achieve the permitted threshold level. Nitrate treatment costs are assumed to decrease each year by 0.5%.

TABLE 5: Nitrate concentration

	Scenario 2.1	Scenario 2.2	Scenario 2.3	Scenario 2.4	Scenario 2.5
Nitrate leaching (metric tons)					
Town	100.70	89.01	99.93	78.04	100.70
Agriculture	1,114.68	611.22	611.22	611.22	602.22
Minus pumped	452.05	452.05	452.05	452.05	452.05
Leachate	763.33	248.18	259.10	237.21	250.87
Treated water (MCM)	105.43	1.73	24.18	0.00	26.86
Nitrate concentration in the aquifer (mg/l)					
1st year	63.00	63.00	63.00	63.00	63.00
10th year	76.42	67.37	67.59	67.15	67.44
20th year	89.49	71.73	72.20	71.23	71.90
30th year	100.76	75.63	76.19	74.79	75.75
40th year	110.47	79.11	79.63	77.92	79.08

They were estimated according to the following formula that relates nitrogen concentrations (in mg/l) to average costs (in cents/CM) (Ivanir, 2000): $C = 0.6356N + 8.6475$. Other costs were estimated as follows: groundwater- \$0.17/CM, water from the national carrier- \$0.25/CM, and treated wastewater- \$0.16/CM. Accordingly, we calculated the treatment costs and total costs of water supply to town and region, as presented in Table 6.

The timing of water treatments is presented in Table 6. In scenario 2.1 water treatment begins at the 21st year, with 15.13% of the groundwater being treated. Later, in the 30th, 40th and 50th years, 44, 58, and 65%, respectively, of the groundwater is treated and diluted with other sources supplied to the town. In Scenario 2.3 water treatment begins in the 29th year, in Scenarios 2.2 and 2.5 in the 50th year; and in Scenario 2.4 no water treatment is needed during the 50 years. Wastewater irrigation as presented in Scenario 2.1 postpones the need for water treatment to the 21st year, i.e., by 8 years as compared with the other scenarios. Water supply costs were estimated in two ways: current cost with interest rate 0%, i.e., the importance to future generations is the same as that to the present population; and capitalized cost was computed with an interest rate of 5%. Derived costs are given in Table 6.

For scenario 2.1 the total amount of treated water is 105.14 MCM, the total treatment cost to the whole region is \$19.40 million without discount (\$0.184/CM) and \$3.27 million at a 5% interest rate. The total cost of supplying water to the region is \$187.68 million without discount (0.178 \$/CM) and \$66.76 millions with discount.

The whole region's water supply costs amount to \$187.68 million in Scenario 2.1 (including wastewater irrigation) as compared with total costs in other scenarios ranging from \$204.16 to \$207.03 million in current values.

TABLE 6: Costs of water supply

	Scenario 2.1	Scenario 2.2	Scenario 2.3	Scenario 2.4	Scenario 2.5
Treatment percentage (%)					
14th year*	0.00%	0.00%	0.00%	0.00%	0.00%
20th year	0.00%	0.00%	0.00%	0.00%	0.00%
30th year	44.08%	0.00%	4.22%	0.00%	3.01%
40th year	57.94%	0.00%	20.16%	0.00%	18.88%
50th year	65.34%	28.84%	30.19%	0.00%	28.91%
Costs ($ millions)					
Current treatment costs	19.40	0.27	3.81	0.00	4.24
Discounted treatment costs	3.27	0.02	0.49	0.00	0.55
Supply for town	113.88	109.82	109.08	111.78	106.55
SUPPLY FOR AGRICULTURE	73.79	95.25	95.25	95.25	97.61
Current total costs	187.68	205.07	204.33	207.03	204.16
Discounted total costs	66.76	72.25	75.23	69.43	74.17

* Since Table 5 begins from the time water passes the unsaturated zone, the 1st year in Table 7 is the 14th year in Table 6.

Water treatment costs are the highest in Scenario 2.1 (current cost of $19.40 million) as compared with $0-4.24 million in the other scenarios. Discounted treatment costs are $3.27 million in scenario 2.1 as compared with $0-0.55 million for the other scenarios. The annual increase in treatment costs is $0.14-0.17 millions (multiplied by a capital return coefficient of 0.054 for 50 years and an interest rate of 5%). This can be divided by the wastewater quantity (9.08 MCM), which means the average annual increase in treatment costs is $0.015-0.017/CM.

The total capitalized discounted cost in Scenario 2.1 is $66.76 million, which is lower than those in the other scenarios ($69.43-75.23 million). Hence, wastewater irrigation (scenario 2.1) is the cheapest alternative, because of the low costs of treated effluents, although it requires water treatment processes to be initiated earlier. It should be noticed that this decrease in total costs is accompanied by an increase in groundwater nitrate concentration.

This paper compares wastewater treatment with other scenarios, which differ in their water supply sources, in order to compare profitability and applicability of various treatment processes, as influenced by the combination of water supply sources. Further studies will extend the nitrate-leaching model to include the nitrate balance for various scenarios.

5. Summary and conclusions

Wastewater may serve as an important water source for irrigation under conditions of water scarcity. However, its use may affect agricultural yields and profits, and also future groundwater quality. These effects can be represented by the impacts of salinity and nitrogen constituents. Since wastewater irrigation increases groundwater pollution, adequate treatment processes should begin earlier, to ensure a supply of good-quality drinking water. Environmental effects of wastewater irrigation can be estimated in terms of the increased water supply costs and the enhanced groundwater pollution.

Regarding the economic effects of wastewater irrigation on groundwater, we have developed a schematic method by which to compare several scenarios that combine various water sources to supply a town and agriculture. This model was applied to examine the effects of accelerated contamination with chlorides and nitrates. The model can be extended to include other regions and scenarios, in order to assist decision makers in understanding and planning water supply sources and treatment processes, as influenced by irrigation with treated effluents.

This model is applicable to other countries facing the need to supply recycled wastewater for irrigation while preventing undesirable impacts – mainly those relating to accelerated pollution of groundwater – while minimizing water supply and treatment costs.

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THE ENVIRONMENT SECTOR IN JORDAN
Some Key Issues and Needs for Risk Assessment

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1. Introduction

Jordan is located in the eastern region of the Mediterranean, it has an area of about 90 000 Km2. More than 70 % of the area is desert (Badia). Precipitation rate ranges from less than 50 mm in the Badia to 650 mm in the highlands. Jordan's population has increased noticeably during the past 5 decades. The population reached 5 millions in 2000, the population growth rate is quite high (4 %). About 60% of the Jordanian people are less than 24 years of age. Education is at high standards; Jordan has 7 public universities and ten private universities.

With regard to natural resources, Jordan suffers from scarce resources of water and energy, but it has good reserves of minerals such as phosphate, potash, limestone, oil shale, in addition to Dead Sea minerals. Jordan has started attracting international investment in different sectors. It has established a number of industrial states and free zones. Aqaba, the only sea connection in Jordan, became a Special Economic Zone, where industrial, touristic and commercial activities exist and would expand.

2. Environmental Management

During 1980 – 1995, environmental affairs used to be managed and coordinated through the Department of the Environment at the Ministry of Municipalities, Rural Affairs, and The Environment. The Department had also the task of preparing a national environmental strategy in cooperation with concerned parties. The strategy was finalized in 1991 and called for a comprehensive legal framework, institutional building, education and promotion of public awareness.

In 1995, the Environmental Protection Law was ratified, by which the General Corporation for Environment Protection (GCEP) was established. The Law gave GCEP the mandate and responsibility of managing environmental protection in Jordan. GCEP reports to the Environmental Council headed by the Minister of Municipalities, Rural Affairs, and The Environment.

To ensure efficient and comprehensive approach to environmental management, His Majesty King Abdullah II has advised the government to initiate arrangements for the establishment of a separate ministry to manage the protection of the environment in the country.

I. Linkov and A. Bakr Ramadan (eds.),
Comparative Risk Assessment and Environmental Decision Making, 381–384.

3. Some Key Environmental Issues And Needs

3.1 WATER RESOURCES:

As noted before, Jordan has scarce water resources. Per capita consumption is around 180 m3 per year, and this amount would decrease if no alternatives were deployed and new resources were exploited.

Current projects / plans include

- Water conveyance from Disi aquifer (south) to Amman.
- Red – Dead Canal.
- Brackish and sea water desalination.
- Improving domestic wastewater treatment.
- Constructing new dams.
- Water conservation and optimization in Agriculture.
- Water conservation at large users
- Improving the water supply network to minimize losses.

Water quality is monitored by the Water Authority of Jordan. Additionally, through yearly contracts the Royal Scientific Society conducts water quality monitoring and assessment for major water bodies in Jordan such as King Talal Dam, and King Abdullah Canal. It also assess drinking water quality and a number of industrial effluents through the National Project for Water Quality undertaken for GCEP.

3.2 CLEANER PRODUCTION AND WASTE MANAGEMENT:

The government is promoting international investment in Jordan. A number of industrial estates and free zones were established in different parts of the country. It is also Jordan's plan to ensure sustainable development, and therefore cleaner production and proper handling and management of waste is necessary.

At present a large number of industries exist in the northern, middle and southern regions. Most of the industries are small and medium scale enterprises. Examples of the existing industries include textiles, tanning, metal finishing, steel manufacturing, food and beverages, detergents, paints, plastics, batteries. Major industries include phosphate mining and fertilizers production, potash, and cement.

Jordan is promoting better utilization of mineral resources. It has started to expand in the production of fertilizers, potash products, and Dead Sea minerals rather than exporting raw materials only. There are also plans for the utilization of oil shale resources in the country.

As expansion in the industrial sector is sought, cleaner production and waste management have also to be ensured to avoid and minimize negative impacts on the environment. Pollution prevention at source, industrial wastewater treatment, air emissions treatment and control are required for some of the existing industries and certainly for the planned ones. It should be noted that major industries have already started implementing environmental management system (ISO 14001).

3.3 SOLID WASTE MANAGEMENT

Through out the past five decades, the number of solid waste landfills have increased to 24 distributed in the cities and municipalities. The management of solid waste handling is in need for upgrading. Waste segregations and recycling has started in some activities through programs conducted by NGOs and by the Greater Amman Municipality, this practice ought to be enhanced and disseminated among the public through effective programs. On the other hand, more work has to be directed to improving the method of disposal of solid waste and the design and operation of the landfills.

3.4 HAZARDOUS WASTE MANAGEMENT

Hazardous waste has been of concern for the past few years, previous work in 1994 identified major sources and types of hazardous waste in Jordan and provided a proposal for handling hazardous waste. GCEP is at present working on the preparation of a hazardous waste facility, however a comprehensive and up to date hazardous waste management plan is still needed.

3.5 AIR POLLUTION

Air pollution is of concern in the industrial areas and in Amman city center where high traffic and commercial activities exist. Ambient air quality standard was prepared in addition to standards for emissions from stationary sources. Air quality monitoring is being undertaken for major industrial sites by the Royal Scientific Society for GCEP. Comprehensive and continuous monitoring of air quality in the industrial areas would become necessary in the future.

4. Needs For Risk Assessment

As Jordan is striving for expansion in new developments and in managing environmental issues, the planning process has to be fostered by adequate risk assessment (RA)studies.

The surfaced needs for risk assessment include use of RA tools in environmental management such as in environmental impact assessment, in sites rehabilitations, in selecting the location of certain treatment facilities (solid waste landfills, chemical storage areas), in transportation; accidental.

Currently simple methodologies are being applied in environmental studies, matrices for assessment based on likelihood, level of impact, frequency of occurrence are used, these are supported by simple modeling of air pollution and water pollution. As information technology is being developed and utilized in environmental applications, it is expected that other supporting tools such as GIS and databanks would also be in operation to influence the application of RA. Major focus is being now given to the management of hazardous substances, in terms of documentation, storage, transportation, use, and disposal. It is anticipated that an information management

system will be established and that RA would be an integral part of the system. The system ought to connect the concerned authorities and to manage data, records, permits, etc to avoid any duplication or gaps in the management processes. It will also be used to monitor and assess handling practices.

Another important area where RA is to be applied is in complementing the current assessment activities of water quality, soil quality, and air quality. As with such assessments RA can be an effective tool in decision making concerning new development plans and projects. This would include siting new development projects, master planning of industrial areas, and responding to any environmental incident.

At present, awareness and capacity building is needed to enable application of RA as an integral tool in decision making. This can be done through well designed programs targeting all stakeholders; decision makers, technical staff, and the environmental organizations. An important stakeholder is also the industrial sector.

5. Conclusions

- At present risk management is being used in different levels, support in advancing risk management is needed.
- Risk management is an important tool in the assessment and management of environmental issues such as water quality, air quality, waste management
- The planned development in the economic sectors ensuring sustainable development requires risk management tools to be imbedded in decision making.
- To be able to move forward with risk assessment at decision making level, special training and awareness is needed. The use of IT should also be enhanced alongside.

COMPARATIVE RISK ASSESSMENT FOR HOMOGENEOUS AND NONHOMOGENEOUS MAMMALIAN POPULATIONS EXPOSED TO LOW LEVEL RADIATION

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Abstract

We have developed mathematical models that describe radiation-induced mortality dynamics for homogeneous and nonhomogeneous (in radiosensitivity) mammalian populations. These models relate statistical biometric functions with statistical and dynamic characteristics of a critical body system in specimens belonging to these populations. The model of mortality for the nonhomogeneous population involves two types of distributions, the normal and the log-normal, for its specimens with respect to the index of radiosensitivity for critical system cells.

The mortality model for the homogeneous population quantitatively reproduces the mortality rate of laboratory mice chronically irradiated at low dose rates when the hematopoietic system (specifically, the thrombocytopoiesis) is the critical one. Comparison of the results obtained within the framework of the mortality models for homogeneous and nonhomogeneous populations show that the mortality model for the nonhomogeneous population predicts a higher mortality rate and a lower survival than would have been predicted from the averaged values of the radiosensitivity index of the critical system cells. The level of chronic dose rates presenting a hazard to nonhomogeneous mammalian populations becomes lower as the variance of their radiosensitivity indices become greater. For individuals possessing hyperradiosensitive critical system cells, even low-level irradiation can lead to mortality. These modeling results demonstrate the importance of taking into account the variability of individual radiosensitivity when predicting the mortality of mammals exposed to low-level irradiation.

These models of radiation-induced mortality, as well as the approaches suggested in the course of their elaboration, outline new pathways in the development of radiation risk assessment methodology. Additionally, the same methodology produces other useful information: a criterion is established that elucidates the groups of radiation risk among population residing in areas with elevated radiation background and among persons subjected to occupational irradiation. (Only routine blood sampling is necessary.) Applying the complete set of preventive and protective measures to persons revealed enables one to reduce the radiation risk both for the individuals and for the population as a whole.

I. Linkov and A. Bakr Ramadan (eds.),
Comparative Risk Assessment and Environmental Decision Making, 385–392.

1. Introduction

An urgent problem for radiological protection is ensuring the safety of populations in areas with an elevated radiation background. To resolve this problem, it is necessary, first of all, to develop new approaches to radiation risk assessment. Current risk-estimation methods, as noted in [1], are not always applicable to low-level chronic exposures because of ambiguity of chronic radiobiological effects. Therefore, new approaches must not ignore the intrinsic properties of the irradiated organism. The implementation of such approaches requires the development and investigation of mathematical models describing mortality as an ultimate result of the radiation-induced damage of mammalian organisms. It is this objective that the present paper is devoted to.

2. Mathematical Model of Mortality for a Homogeneous Population

Initially, we develop here a mathematical model that describes radiation-induced mortality dynamics in a homogeneous (with respect to radiation effects) mammalian population. The radiobiological concept of the critical system [2] forms the basis of our model. According to this concept, the principal cause of radiation-induced death in mammals is failure of one of the organism's vital systems, which manifests itself in the disruption of cellular kinetics and a decrease in the number of functional cells of the particular system below the level required for survival. For each of the studied doses and dose rate intervals there seems to be a specific critical system whose damage will determine the mechanism of radiation sickness and eventual death of the mammals.

Also used in our model is the stochastic approach proposed by Sacher [3], who modeled a homogeneous population in which every individual had the same average values of all physiological variables and their fluctuation parameters. Sacher described this population using a random variable that served as a generalized index of physiological state, and when this variable reached or exceeded a critical level, he regarded the situation as analogous to mortality.

According to the critical system concept, we choose the deviation of the concentration of critical system functional cells from the normal level as an index of physiological state, and we assume that reaching or exceeding a threshold value by this amount is a death analog. In this way a model is created that relates the statistical biometric functions (mortality rate, probability density, and life span probability) with the dynamics of the concentration of critical system functional cells and with the statistical characteristics of this physiological index in the mammalian species in question [4, 5].

The model is used to simulate the mortality of mice exposed to chronic radiation in the range of the low dose rates responsible for bone marrow syndrome. In this case hematopoiesis (namely, thrombocytopoiesis) is the critical system. The concentration dynamics of its functional elements, thrombocytes, are calculated by a specially developed dynamic model [4, 5]. There is qualitative and quantitative agreement between modeling results and experimental data [6] regarding the mortality rates of LAF1 mice exposed and not exposed to low-level chronic irradiation (Fig. 1).

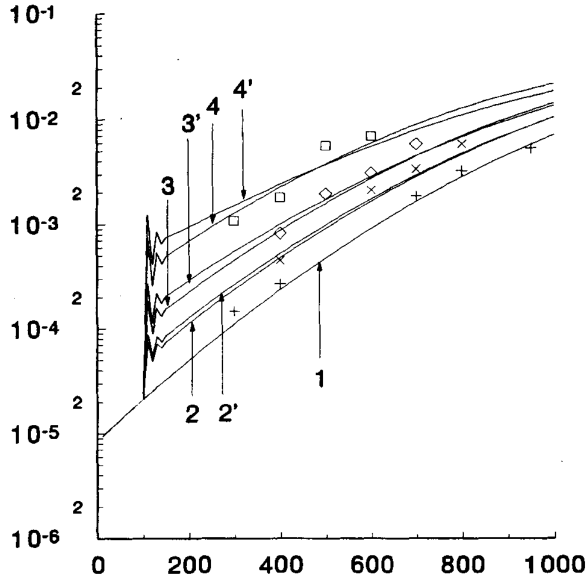

Fig 1. The modeled mortality rates of homogeneous and nonhomogeneous (normal distribution, high variance in radiosensitivity) populations of mice not exposed (curves 1 and 1') and exposed to chronic irradiation at dose rates of 0.022 Gy/day (curves 2 and 2'), 0.044 (curves 3 and 3'), and 0.088 Gy/day (curves 4 and 4'). The symbols (+), (X), (◊), (□) indicate the corresponding experimental data on the mortality rate of LAF1 mice [6]. The abscissa shows the age of animals in days; the ordinate the mortality rate in 1/day.

A major difference and advantage of our model as compared to others is that identification of its coefficients does not require data on mortality dynamics of irradiated mammalian populations. Only data on the population's mortality in the absence of radiation are needed, as well as a limited number of experimental or clinical observations of the behavior of the respective critical system under acute or chronic irradiation. Therefore, this model can be employed to predict the mortality dynamics of large mammals and humans under low dose rate chronic irradiation, the duration of which is commensurable with the life span.

3. Mathematical Model of Mortality for a Nonhomogeneous Population

Next we extend our model of radiation-induced mortality to nonhomogeneous populations. We start from experimental studies suggesting that populations of various mammalian species, including humans, contain a small proportion of specimens (from 10 to 20%) that show hyperradiosensitivity [5, 7, 8]. Therefore, our approach implies the importance of taking into account the nonuniform radiosensitivity of individuals in

a population. Accordingly, our model of radiation-induced mortality dynamics for a nonhomogeneous (in radiosensitivity) population [4, 5] is based on the assumption of nonuniform individual radiosensitivity of critical system precursor cells.

The distribution of individuals in the radiosensitivity index is described by a continuous function. An important component of the model is an adequate approximation of the continuous function by a discrete function. This transition from a continuous distribution to a discrete one is equivalent to the representation of the initial nonhomogeneous population as a set of a finite number of homogeneous subpopulations. The radiosensitivity index of the critical system cells in individuals of each homogeneous subpopulation, and also the number of these individuals, is uniquely determined by the initial continuous distribution.

Another important component of the model is the set of formulae used to express the biometric functions describing the mortality dynamics of the nonhomogeneous population; this is done using the biometric functions that define the mortality dynamics of the constituent homogeneous subpopulations. To calculate the radiation-induced mortality dynamics of the subpopulations, we use the mathematical models of mortality dynamics for the homogeneous population and of the respective critical system.

The resulting structure of the model reflects actual occurrence of adverse radiation effects in mammals. The first level is that of a critical system, whose radiation injury is largely determined by the radiosensitivity of its constituent cells. Second is the level of the whole organism: here the probable outcome of irradiation depends mainly on the extent of radiation injury to the respective critical system, *i.e.*, on the individual cell radiosensitivity of this system. The third level is at the scale of the population, which includes animals having critical system cells of differing individual radiosensitivity. It follows then, that our model of mortality is essentially a mathematical description of the cause-effect relationships that develop over the course of radiation injury to mammals.

We used this model study the effect of low-level chronic radiation on the mortality of nonhomogeneous populations of mice. We employ two types of distributions, the normal (Gaussian) and the log-normal (both frequently encountered in biology), to describe the distribution of nonhomogeneous individuals as described by the radiosensitivity index of the critical system precursor cells (thrombocyte precursors). Then, in model experiments, we investigate the dependence of population mortality on both the type and the variance of the distribution of individuals' radiosensitivity indices.

To compare model predictions for nonhomogeneous and homogeneous populations, we take the mean value of the radiosensitivity index of the critical system precursor cells for specimens of the nonhomogeneous population to be equal to the value of this index for specimens of the homogeneous population. Additionally, we simulate the mortality of nonhomogeneous populations of mice exposed to chronic irradiation as occurring at the same dose rates as the mice of the homogeneous population (Figs. 1, 2).

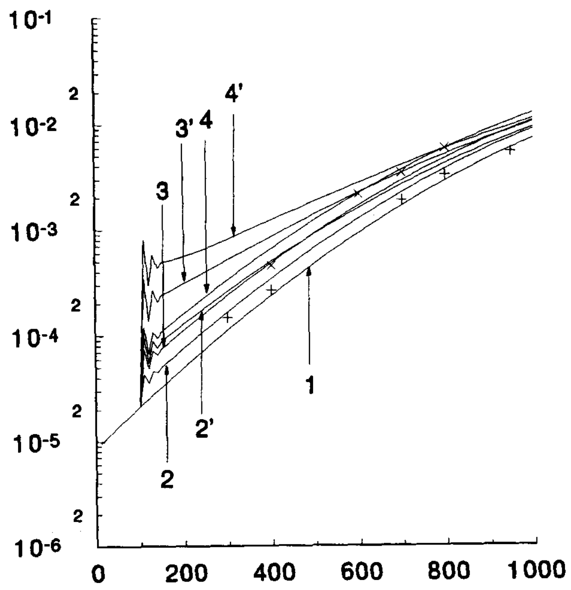

Fig 2. The mortality rate of homogeneous and nonhomogeneous (log-normal distribution, very high variance) populations of mice not exposed (curves 1 and 1') and exposed to chronic irradiation at dose rates of 0.011 Gy/day (curves 2 and 2'), 0.022 Gy/day (curves 3 and 3'), and 0.033 Gy/day (curves 4 and 4'). The symbols (+) and (X) *indicate the corresponding experimental data on mortality rate of LAF1 mice not exposed and exposed to chronic irradiation at a dose rate of 0.022 Gy/day [6]. The abscissa shows the age of animals in days; the ordinate, the mortality rate in 1/day.*

Comparative analysis of the model results shows the following: consideration of both the normal and log-normal distributions in individual radiosensitivity indices results in higher modeled rates of radiation-induced mortality and a lower survival than would have been predicted from the averaged (point estimate) indices alone.

In addition, differences in prediction are more pronounced when there is greater scatter in the individual radiosensitivity indices of a nonhomogeneous population. These differences are the greatest when the specimen distribution in the nonhomogeneous population is log-normal with a high variance (Fig. 2).

These results suggest that the chronic dose rates that present a certain risk for nonhomogeneous mammalian populations lowers as the scatter of values of individual radiosensitivity indices of a modeled population increases. For animals having hyperradiosensitive precursor cells, even low-level radiation can have fatal consequences. Obviously, these model results have considerable theoretical and practical importance.

4. Identifying Hyperradiosensitive Individuals

Analysis of clinical data on irradiated humans [5] supports the validity of the principal concepts forming the basis of our model of radiation-induced mortality of nonhomogeneous mammalian populations, and indicates that these concepts must be taken into account in modeling radiation effects on human populations. The finding that 10 to 20% of humans have enhanced radiosensitivity is quite significant [5, 8]. It supports our conclusion, discussed above and drawn from the results of our above-described modeling experiments, that even very weak irradiation can have fatal consequences for individuals who have hyperradiosensitive critical system precursor cells.

All this provides evidence that identifying hyperradiosensitive individuals, and making them the first priority when applying the complete set of preventive and protective measures, enables one to reduce the risk of mortality both for these people and for the population as a whole.

By analogy with epidemiological terminology, the subpopulation of hyperradiosensitive individuals can be called "the group of radiation risk." The task of singling out this group among the population is not trivial. For instance, some authors [9] proposed that radiosensitivity be predicted according to indices showing the organism reactivity under normal conditions and in the presence of adverse non-radiation factors. This technique seems to be promising and should be further examined. In our opinion, however, it is more suitable for people who are expected to experience radiation exposure and less suitable for populations residing in contaminated areas. The fact is that the response of an organism that experiences chronic irradiation at low dose rates has already been altered, and an additional exposure to adverse non-radiation factors can lead to misleading results.

We propose a safe, simple, and inexpensive method of identifying the radiation risk group among populations in areas having an elevated radiation background. The method is based on the radiobiological concept of critical systems [2] and is as follows. In the range of dose rates typical of most contaminated areas, the critical system of the human organism is the bone marrow blood-forming system. Consequently, in such areas the radiation risk group should include individuals whose bone marrow blood-forming precursor cells show hyperradiosensitivity. Taking samples of bone marrow cells is not a harmless procedure. Therefore, direct determination of radiosensitivity of bone marrow blood-forming precursor cells, rather labor-consuming in any case, cannot be the routine method for identifying persons with elevated radiosensitivity, though it can be performed in extraordinary cases.

There exists, however, a method of radiosensitivity assessment for bone marrow precursor cells from indirect data. As shown by model calculations and experiments [5, 10], chronic irradiation with low dose rates brings about new and abnormal concentrations of blood cells—thrombocytes, lymphocytes, erythrocytes, and granulocytes. As a consequence of chronic irradiation in certain ranges of low dose rates, the new stationary concentrations of bone marrow precursor cells in the lymphopoiesis, granulocytopoiesis, and erythropoiesis systems and even of functional cells (granulocytes) may exceed the normal level. Therefore, concentrations of lymphocytes, granulocytes, and erythrocytes in the blood of mammals exposed to

radiation at low dose rates cannot serve as an adequate characteristic of radiosensitivity of precursor cells of these systems. However, analysis of the thrombocytopoiesis model has revealed the following picture: new stationary concentrations of bone marrow precursor cells of this system and of thrombocytes in the blood of mammals exposed to chronic irradiation at any dose rate are always below the normal level. Furthermore, at a constant dose rate, the higher the radiosensitivity of thrombocyte precursor cells in the bone marrow, the lower is the new stationary concentration of thrombocytes. In view of the fact that, in many cases of bone marrow syndrome, it is the failure of the thrombocytopoietic system that is responsible for the death of mammals, we believe that the thrombocyte blood concentrations can serve as a reliable indicator of bone marrow precursor cell radiosensitivity.

Thus, it can be expected that, in a radiation-contaminated area with a nearly uniform (but increased as compared to normal) radiation background, hyperradiosensitive individuals will have lower thrombocyte blood concentrations than the average for the population of this area. Hence, routine blood sampling for thrombocyte concentrations, and subsequent simple calculations to find the average thrombocyte concentration for a homogeneous cohort of people, are sufficient to identify hyperradiosensitive individuals in each cohort. This is just a general idea of the proposed method for finding out persons belonging to the radiation risk group. Clearly, practical application of this method must be preceded by working out, on the basis of available clinical data, some optimal criteria for grouping individuals in cohorts, and by elaborating mathematical procedures or programs for statistical processing of the results.

5. Conclusion

The principal result of the research described in this paper is a new approach to radiation risk assessment and the realization of this approach as a family of mathematical models. These models enable one to predict the effects of low-level chronic radiation exposures on the principal critical system (hematopoiesis) of an individual, as well as on populations whose individuals are nonhomogeneous in radiosensitivity. Comparative analysis of modeling results regarding dynamics of radiation-induced mortality for homogeneous and nonhomogeneous populations of mammals shows that taking account of the nonuniform radiosensitivity index of critical system precursor cells in individuals of the nonhomogeneous population leads to higher mortality rates and lower survival than could have been predicted from the average radiosensitivity index alone. These modeling results suggest that a new strategy of radiation protection must be adopted for populations in areas having an elevated radiation background: identification of and priority for hyperradiosensitive individuals when applying the whole set of preventive and protective measures, including moving them to non-contaminated places of residence. A method of defining radiation risk groups among populations in areas having an elevated radiation background is also proposed. Thus, the comparative study of radiation-induced mortality models for homogeneous and nonhomogeneous populations makes it possible to give a quite specific recommendation for the protection of populations in contaminated regions.

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RISK ASSESSMENT OF THE INFLUENCE OF ANTHROPOGENIC FACTORS ON HUMAN SAFETY AND HEALTH

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Abstract

Technogenic risks in Ukraine may exceed those in other countries. Besides, the largest technogenic catastrophe in the world occurred in Ukraine—the Chernobyl disaster.. Therefore, the estimation of endogenic and exogenic risk is extremely important, and not just for Ukraine. Our calculations were conducted using modified Gomperca-Meykema models developed by Russian scientists [2]. This work describes the major results of our quantitative estimation of the exogenic (i.e., external, unconnected with internal biological causes) and endogenic (i.e., caused by biological and/or genetic factors) risks of death, or predicted death rate of the Ukrainian population.

1. The Mathematical model of the estimation eczogenic and endogenic forming risk

The Gomperca-Meykema Law corresponds to the principle of versatility. It describes the distribution of life expectancies for different organisms, including humans. Studies have shown that thishis law describes population's death rate (death-rate coefficients) for any country at any time periods [1, 3, 5].

The Gomperca-Meykema equation for endogenic and exogenic risks [1, 4] can be written as:

$$d(\tau, t) = R(t) + \beta\exp(\alpha\tau), \tag{1}$$

where: $d(\tau, t)$ describes death rate coefficients of the population; t represents variable time; τ is a person's age; $R(t)$ is a variable that depends only on time and influence exogenic risk of death; $\beta\exp(\alpha\tau)$ depends on age and describes endogenic risk and α and β are constant. $R(t)$ was found to be decreasing for developed countries over the past 50 years. Coefiicients α and β remain constant for specific population groups. This allows us to consider $R(t)$ in equation (1) as the exogenic risk and $\beta^{(\alpha\tau)}$ as the endogenic death rate for the specified population group.

Other studies show that $R(t)$ also depends on age τ and, in relation to equation (1), describe exogenic risk of deaths for all age groups. Paper [2] offered a mathematical model for the separation of endogenic and exogenic risk based on population death rates and other statistical information. This takes the form:

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$$d(\tau, t) = A(\tau) + B(t)C(\tau), \tag{2}$$

where: $d(\tau, t)$ the death-rate coefficients for the specific population groups; t - time; τ - an age; $R(\tau, t) = B(t)C(\tau)$ describes exogenic population's death-rate, which depends on age and time; $A(\tau)$ describes the endogenic risk which depends only on age and is described by the Gomperca-Meykema law.

Results of studies have shown that the mathematical model (2) aptly describes death rate intensity (death-rate coefficients) of a population during various periods of time. The endogenic death-rate follows an exponential function, while the exogenic risk follows a linear function.

2. Analysis of death-rate coefficients in Ukraine

For the quantitative estimation of endogenic and exogenic risks, it is necessary to have statistical death-rate coefficients for the Ukrainian population. Authors used data collected by the Ukrainian Government in 1980-2000. It report death rates ffor the following age groups: (0-4), (5-9), (10-14), (15-19), (20-24), (25-29), (30-34), (35-39), (40-44), (45-49), (50-54), (55-59), (60-64), (65-69), (70-74), (75-79), (80-84), and (85 and more) years. The age death-rate coefficients can be derived from the chracteristics of a specified age group as:

$$K_i = M_i / \bar{S}_i \times 1000, \tag{3}$$

where: M_i equals the number of people in the i-y age group deceased over the year, and $\bar{S}_i$ equals the average annual number of people in the i-y age group.

Benchmark analysis of the death-rate coefficients of Ukrainian males and females in the 18 age groups for the years 1980-2000 has shown that the death rate is high for infants (age group 0-4) for both males and females. The death-rate for boys is well above that of girlsfor the 5-9-year age group. However, death-rates for these age groups decrease with age. For the 10-14 age group, the death-rates for both sexes begin to increase, and also become vastly differentiated between the sexes within the same age groups. Figure 1 shows death-rates of the Ukrainian population in 1998 by age group for bothes sex, as well as separately for males and females. For greater clarity, the ordinate axis uses a logarithmic scale. As can be seen in Figure 1, for all age groups the death-rate coefficients for men are much higher than those for women, except for the age group of 85 and greater, for which the difference is smaller.

When studying a question regarding a difference in quantities of deaths of males versus females, comparison of the two is facilitated by the definition of a value that describes the excess of male death-rates above female death-rates. We calculated the absolute value of the difference between age of men and women having an identical death-rate for a given period in time. In Ukraine, in 1998, seventeen-year-old boys the same death rate as 32-year-old women; that is, the absolute value of the ,male of 17 years minus the female age of 32 years equaled 15 years.

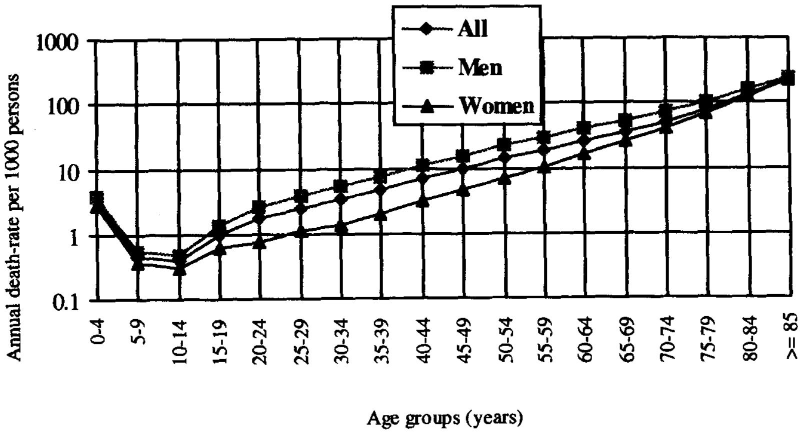

Fig. 1. Death-rate coefficients of the Ukrainian population in 1998

At the age of 22 years, males have the same death rate as 42-year-old women, or a difference of 20 years. As males get older, the difference is reduced, so that for males 42 years old, it is equal to 15 years; for 62-year-old men, 10 years, and for males at 72 years, it is 5 years. Nevertheless, in Ukraine the excess of the male death rate above the female is quite large. The excess of Ukrainian male death-rates above those of females demonstrates the need for the development and implementation of effective measures that will suspend the growth of this difference and minimize the gap by increasing the life expectancies of Ukrainian males.

Our comparative analysis of death-rate coefficients for Ukrainian males and females in 1980-2000 confirm the assumptions of the well-known demographer B.Urlanisa [6], who theorized that death-rate parameters have a biological nature, but upon specific social conditions exert their deciding influence. The age dynamics of the death-rates for two periods: the period of high death-rate for children, when death-rate intensity decreases with age (during early and late childhood, or the 0-4 and 5-14 age groups).

In recent years, the death rate of men in the age groups 40-44, 45-49 and 50-54 years has increased. In 1990 the death-rate coefficients for these age groups were, respectively, 6.63%, 10.59%, and 14.80%. In 1995 they had already increased to 11.39%, 15.66% and 22.89%. These values speak of many accidents among men in different age groups. Even at early ages, boys are involved in risky acts more often than girls, and women are less fond than men of risky kinds of sports, automobiles, and motorcycles.

In the conditions which currently exist in Ukraine, the reduction of the male premature death rate and an increase in the male life expectancy will be possible only after control of new risk factors is established. For this purpose, it is necessary to

increase health protection so as to reduce the level of smoking and alcohol use, as part of a constant campaign for a healthy way of life.

3. Analysis of exogenic and endogenic risks for different age groups

In Figure 2 we present the results of our calculations of death-rate, for all age groups and for males and females separately, as well as both sexes together. They demonstrate that the prominent feature of the death-rates all gender and age groups, from 15-19 years and older, is their tendency to increase.

The axis *X* of Figure 2 displays population age groups, and the ordinate (*Y*) axis shows the death-rate intensity of the population (number of deaths / 1000 of men per one year). For nine age groups, from age group 20-24 to age group 60-64, the size of exogenic component is much higher than the endogenic component of death-rate. For other age groups, a different tendency is observed: the endogenic component of death-rate surpasses the exogenic part. However, for all age groups, endogenic risk component is more than exogenic component. A similar picture is observed for other years. The character of the changes in endogenic component corresponds to the exponential law of increase.

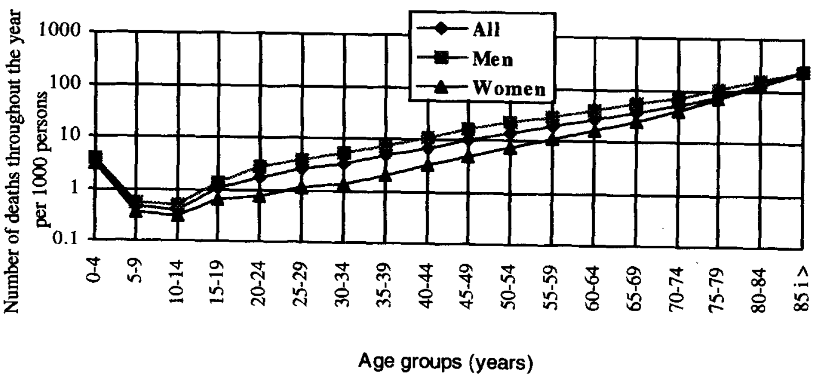

Fig. 2. Distribution ofendogenic and exogenic components of death-rates (due to all causes) of the Ukraine population for all age groups in 1998

Our analysis of the exogenic risk of death-rate for the Ukraine population has shown that the largest exogenic risk falls to the 40-44-year-old age group. That is, the risk of death-rate from external harmful factors in this age group equals 76% in 1998, whereas the risk caused by the internal biological reasons equals 24%. In absolute values, the size of the exogenic risk of death-rate for this age group is 4.4×10^{-3}, and the size of the endogenic risk is $1,3\times10^{-3}$. The exogenic (background) risk for relatively "young" age groups (i.e., ages 25 through 60) is of a higher interest. The background risk for these groups is 60% of the full risk of death-rate in these age groups.

Comparison of the relative values of exogenic risk has shown that, over the majority of age groups, they are much higher for the population of males than for that of females (Figure 3). The differences were relatively small only for the age groups of 20-24 years and after 75 years they differ a little.

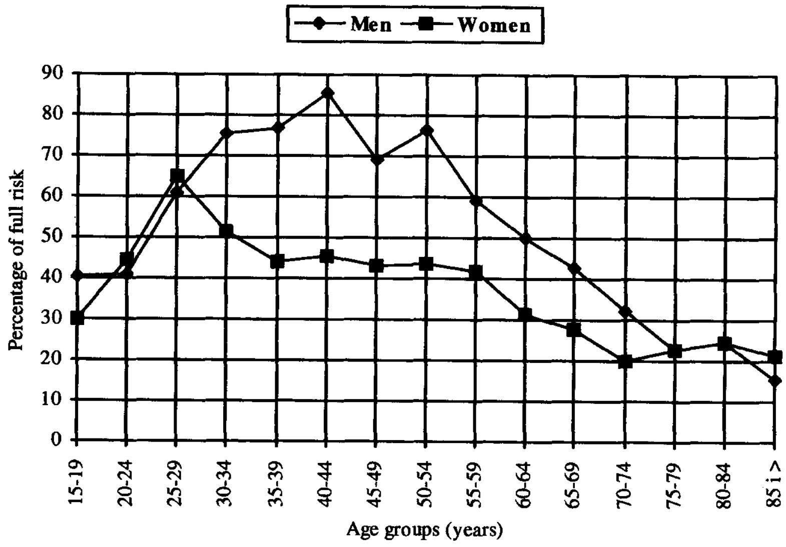

Fig. 3. Relative exogenic risk of death-rate for all reasons, men and women, 1998

Men of ages 40-44 have the highest exogenic risk. The highest exogenic risk for women is seen in 25-29-year-old group. It should also be noted that men in the age groups of 30-34 and 40-44 have the lowest values of endogenic risk component. This fact could be one of the reasons for high death-rates of "young" men (ages 30 through 50) in Ukraine.

In Figure 4 we show the distribution by age group of exogenic risk for all causes, for Ukrainian males and females. For exogenic component, it is acceptable to create relative values of the elements of the vector C_j, which is described in model (2).

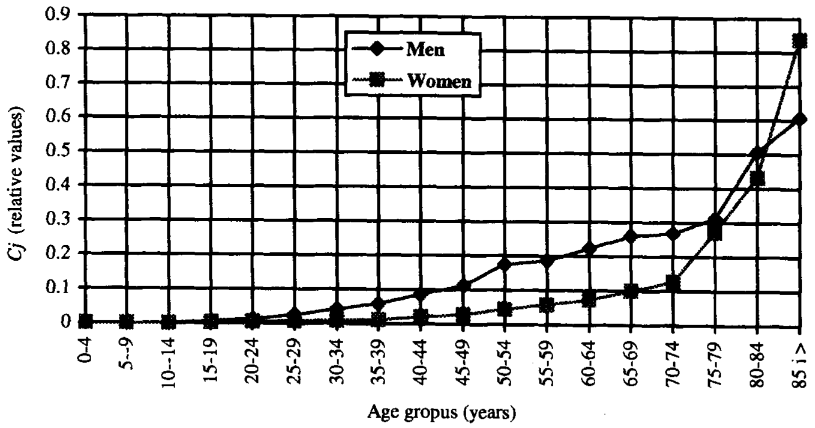

Fig. 4. Exogenic forming of death-rate coefficients for all reasons of the man's and woman's population of Ukraine

It is obvious that from ages 15 through 75, the Ukrainian male exogenic risk is higher than that of females. For a more substantive presentation of the issues, however, it would be necessary to carry out a full analysis, using large amounts of data covering a longer period of time.

4. Dynamics of changes of the exogenic and endogenic risks of the Ukrainian population

The ability to separate exogenic and endogenic components of death-rate of the Ukrainian population have allowed us to calculate exogenic and endogenic risks to the population. Figure 5 shows the dynamics of changes in exogenic risk of death-rate for males, females, and both sexes over the period of 1980 to 2000. The exogenic risk of death-rate for males over the entire time period is much higher than that of females, and it has grown appreciably since 1990, achieving its maximal value in 1995. It should also be noted that the lowest annual death rate in the time period of 1980 to 2000 occurred in 1986, as was reflected in the exogenic risk values. This latter trend is seen for both sexes: the exogenic risk of death-rate for females is also shown to have been increasing since 1991.

In Figure we display the data on the endogenic and exogenic risk of death-rate for male Ukrainians, as a percentage of total risk of fatal cases are given. Since 1986, the male exogenic risk has increased. While in 1990, the percentage of male Ukrainian deaths due to exogenic causes was 38,5%, in 1995 it had increased to 52.1%. In general, for the entire population of Ukraine (both sexes), the percent risk of death-rate due exogenic reasons increased 12.6% from 1990 to 1995, from 32.6% to 45.2%. Over the same period, for males only, this value has increased by 13.5%, whilefor ~females it has increased 11.6%.

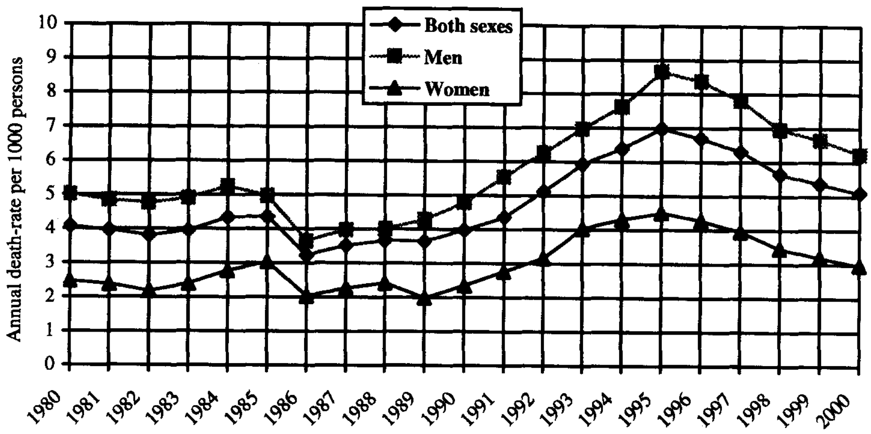

Fig. 5. Changes in the exogenic risk component for males, femalesand all, for all causes

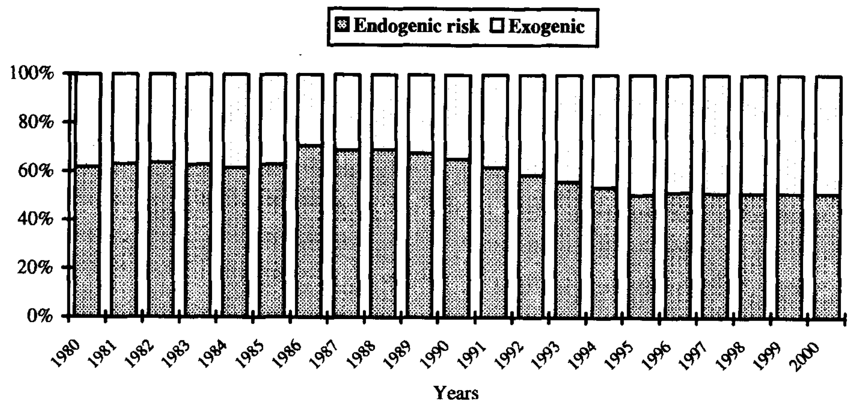

Fig. 6. Exogenic and endogenic risk components for the male Ukrainian population in 1980-2000

In Figure 7, we provide the data on endogenic and exogenic risks as a percentage of total risk for the female population of Ukraine in the years 1980-2000. The general trend of an increase in exogenic risk since 1986 is observed here as well, but the distinct increase does not begin until 1991. The reasons for this substantial growth in exogenic risk for 1991-2000 are connected, primarily with the sharp economic and social changes that Ukraine has seen during that time. The adverse ecological conditions and other consequences of the Chernobyl have also had a considerable effect.

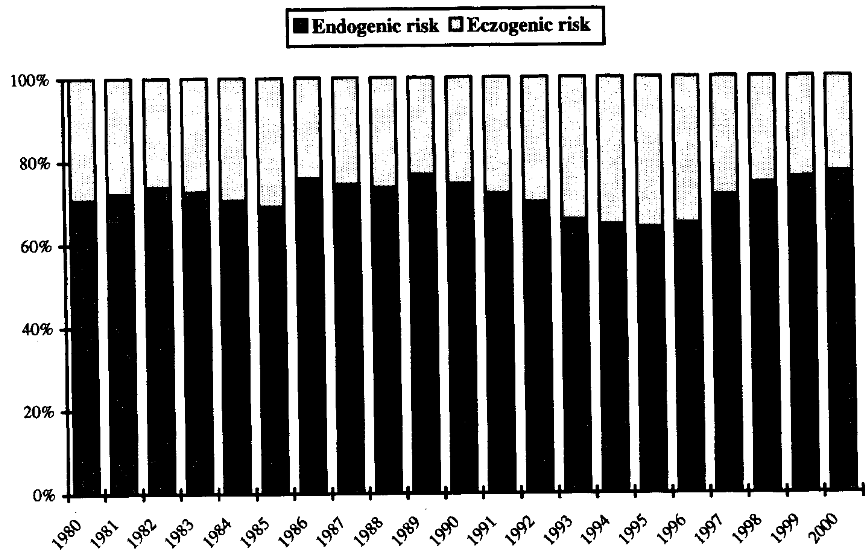

Figure 7. Female Ukrainian population in 1980-2000

Thus, the separate data on exogenic and endogenic risks for males, females, and both sexes of the Ukraine population testify, that, since 1986, the level of fatality caused by exogenic risk has, on the average, constantly risen in Ukraine. However, in 1996 another tendency became visible: the population's level of exogenic component, while still high, has begun to fall. This may be explained by sharp changes in Ukraine'ssocial and economic position during the period of 1986-2000, and changes in people standards of living.

5. Acknowledgement

From the results of our research, we conclude that there is complete justification for the mathematical technique based on a generalization of the ideas that are incorporated in the Gomperca-Meykema law; it has allowed us to reveal the level of influence that external harmful factors exert on the death-rate of the Ukrainian population. Nevertheless, further research is required, particularly on the specification of model factors for age groups under 15 years old.

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ENVIRONMENTAL RISK PREVENTION AND ENVIRONMENT MANAGEMENT IN LITHUANIAN MILITARY LANDS

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Abstract

In the East and West alike, the transfer of former military bases to civilian hands creates challenges for those responsible for redevelopment. The former Warsaw Pact countries and former Soviet Union Republics countries have great difficulties solving the environmental problems that these lands present. The negative effect of military activities on the environment is far from uniform in different military areas. These differences have been determined mainly by the kind of military activity conducted in a particular area, as well as by the individual characteristics of the local natural environment. This report reviews our experience in research on the impact of military activities on the environments of both former and working military sites; environmental risk prevention and control; the Lithuanian experience in environmental management and evaluation of military lands; and ways to address environmental risk reduction, reuse, and renovation of damaged military territories.

1. Introduction

Over the past twelve years the political situation in Central and Eastern Europe has undergone sweeping changes. More than 8000 different military installations all over the world in mixed territories of about 1 million hectares have now become accessible to civil needs [1, 2]. The former Soviet Baltic republics and the Warsaw Treaty countries have been affected likewise. When the Soviet Union withdrew more than one million of its soldiers from these areas in late 1990s, thousands of military bases were closed in the region [3-7]. A whole spectrum of military locations, ranging from isolated computational installations and semi-self-sufficient military communities to well-developed training bases without infrastructure, were left for civilian reuse. [5-10]. The negative effect of military activities on the environment is far from uniform over different areas. This has been mainly determined by the kind of military activity conducted in a particular area, as well as by the peculiarities of the natural environment [22]. Due to the demilitarization process throughout the world, military bases are being closed, and therefore areas of direct military use are becoming smaller. However, after military activity has been terminated, it is necessary to clean up and restore the area before giving it over to civil needs [13,16,19]. In addition, the environmental situations in areas still operated by militaries must be also be carefully supervised [21, 23].

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2. Specifics Impact of Military Territories

There is no data on the exact area that is currently being used for military purposes worldwide. It has been said that 13 best-developed countries in the world (except Russia) have allocated 1.5 million km^2 of their lands for military purposes, which makes up more than one per cent of their total area. Very large areas serve as protective buffers around military installations, or are needed at the time of military exercises [12].

The impact of military chemical pollution on the environment is manifold [11, 20]. It is caused by everyday military activities, training, and the production and testing of weapons and ammunition. It has been estimated that over 260 different chemical substances are released into the environment due to military activities, and that military activities account for 10 - 30 % of the chemical degradation of the environment [13]. Activities on military grounds also have a significant and manifold (though mostly negative) non-chemical impact on the environment [13, 14, 21]. The most evident effect is physical degradation of soil. Also, the building of military bases entails construction work on an enormous scale, especially the construction of underground installations [20]. Due to all of these influences, the disturbed soil starts eroding. By far the greatest physical impact falls on tank training areas and bombing ranges. There, a soil layer several meters deep is violently disturbed by explosive power and machinery, and the area turns into a wasteland. Areas beyond the boundaries of military bases are not immune to explosions either [1, 10, 11].

The specific purpose of military grounds—the use of special technical equipment and machinery—facilitates the presence and formation of specific pollutants—remnants of ammunition and explosives. The production of these materials often a number of uses heavy metals and other hazardous substances, including as copper, zinc, lead, mercury, uranium, bromine, phosphorus, etc. [5, 9, 12, 13, 14], as well as various carcinogenic and toxic chemical compounds This makes these materials very dangerous for the environment and for human health as well [12, 16, 17]. The remnants of these substances penetrate into the soil and, interacting with air and water and affected by the constantly changing physical-chemical environmental conditions, start melting, disintegrating and forming new compounds, These often filter into and are spread by groundwater, thus polluting the areas far beyond the boundaries of military grounds [8].

3. Assessment of Environmental Damages and Risk on the Former Military Lands in Lithuania

The Soviet army left Lithuania in 1993, leaving about 500 military installations, including 277 Soviet military bases (Fig. 1). Their sizes varied over a rather broad scale–from less then 100 m^2 to nearly 14000 ha. In total, these military sites occupied 67762 ha, or 1.04% of Lithuania's territory. Currently, 16.7 % of these lands have been left to satisfy the needs of the Lithuanian military, and the rest has been transferred to civil users [8].

These military sites have served specific purposes - from the establishment of military settlements and military forestry areas to shooting grounds and military airfields (Table 1). This has significantly affected the scale and character of the environmental pollution and destruction at these locations.

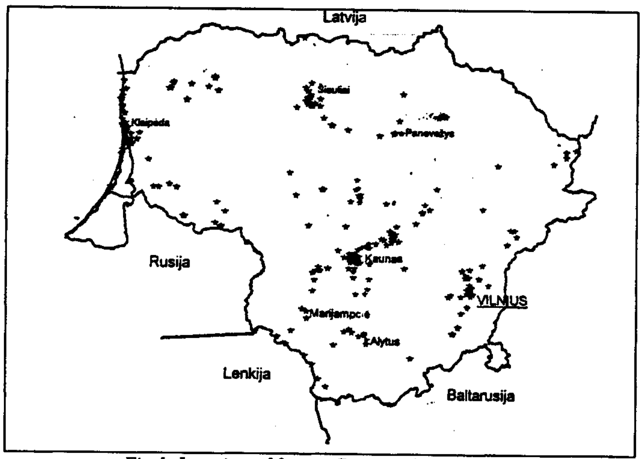

Fig 1. Location of former Soviet military bases in Lithuania

TABLE 1. The distribution of military sites according to their destination

Type of military site	**Number**
Motor-rifles units	3
Landing-party units	10
Artillery units	4
Engineering, transport, railway, supply and building units	33
Airfields, aviation units	15
Storage of oil products and rockets fuel	4
Rockets and antiaircraft bases	31
Warehouses	21
Communication units	35
Grounds and shooting-ranges	12
Border troop units	20
The military infrastructure (settlements, schools, hospitals, shops, military forestry, military tribunals)	63
Training and teaching centres	6
Repairing enterprises	4
Tank units	1
Unites of other types	15
Total:	**277**

When Lithuania took over the Soviet military bases, an Evaluation Committee comprising local specialists was established to evaluate their environmental situations. One of its main tasks was to identify effective measures, which applied, could prevent further spreading of pollutants. Lithuanian experts recorded 2743 sources of actual

pollution in the former military lands. Only 14% of all former Soviet military sites did not contain pollution sources. However, the remainders of 200 types of poisonous chemical substances have been found in the remaining 86% of the sites. Many different flammable materials were also left behind. Ruins of former buildings and other sources of potential danger are present at almost every site.

As we can see from Fig. 2, pollution caused by oil products and rubbish-heaps predominated, as did physical damage to landscapes and soils. Streams of wastewater and wind have transported heavy metal particles, oiled dust, bitumen, and other break-up products from rubbish-heaps to the cleaner areas. Natural ecosystems at these restricted military sites have been severely affected by long-term military activity, and they cannot be easily restored, sometimes causing harm to the health of local residents. Therefore, the main problem surrounding military site reuse at this time is the prevention of further spread of pollution. More specifically, we must achieve immediate localization and liquidation of those pollution sources which pose direct risk to human health and the environment.

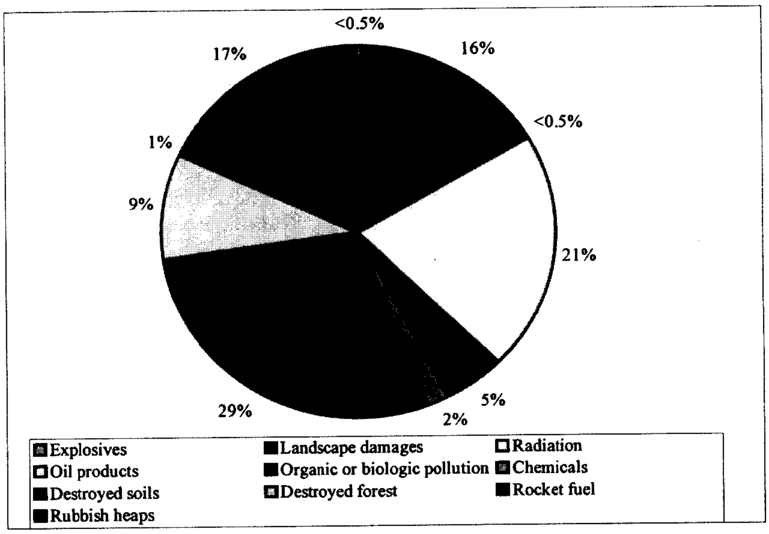

Fig. 2: sources of the actual pollution in former military lands

Based on the research on the environmental situation at former military sites, these areas have been grouped according to their level of investigation and the necessity of preventative measures, clean-up and restoration. Every site has been placed in a category according to the listed criteria. The categories are described by the lettered indexes A, B, C, D, E, F (from the most to the least severe damages). Every site has received an index consisting of two letters. The first letter describes the level of investigation and the necessity of further research and/or preventative work and cleanup; the second letter represents the level of landscape destruction and the necessity

of restoration works. Only one former military territory has been placed in an A category; however, about 80% of sites have been placed in the B, C and D categories.

4. Biodiversity and Conservation values in the Former and Working Military sites in Lithuania

The largest sites—ranging from 2000 to 22000 ha—had been occupied by grounds that, together with defense zones, belonged to six military forest grounds. Military grounds and troops occupying large territories were located in various ecosystems, but most often in sparsely settled, marshy woodlands.

The activities of military sites and their troops were closed to the public. In many respects, including environmental protection, the Soviet military was independent of the Lithuanian government. Thus, there was no possibilities of inventorying the state of its military installations, assessing the natural resources that should have been under protection, or defining their status and the protection measures they needed.

As described above, intensive military activities at these sites have caused great harm to the environment at former military locations: many forests were cut, large areas of soil damaged, water bodies polluted. Nevertheless, due to their closely protected and isolated military status, these sites also experienced conditions that favored the survival of natural ecosystems. Research on this topic is carried out by the author, sponsored by the MacArthur Foundation.

Natural resources inventories of former and currently operating military sites, together with established protective status for such sites where applicable, will create a network of protected areas in Lithuania that can then form a base for the restoration of damaged and degraded military sites. For working military grounds where militaries are training, coordination of military activity and environmental protection will allow the best results.

The principal conclusion we draw from the survey of these closed territories in past and present years is that in spite of the impact of military activity on the environment (soil contamination, soil and slope erosion, landscape degradation) these areas enjoy very rich biodiversity because of their limited access to humans. This situation is typical for all of the military grounds investigated by this survey [21, 23]. (On a wider scale, a striking contrast between environmental degradation and rich wildernesses in close proximity is a common feature throughout Eastern Europe) During these investigations of former and working military sites, we have found significant natural resources that urgently need to be protected.

5. Actions on the Environmental Risk Prevention in Former Military Lands

Regardless of how these military sites will be utilized in the future, the following environmental risk prevention measures are being applied:

a) removal of radioactive pollution sources;
b) removal of explosives;
c) isolation and removal of oil pollutants;

d) removal of aggressive waste and scrap; and
e) neutralization and removal of chemicals.

These measures are meant to mitigate the danger arising from pollution migration and to stop dispersion of explosives or other dangerous materials. They are also meant to guide the environmental protection systems that will be established in these territories. The first preventive action, which has been performed at all applicable military sites, was the removal of all sources of radioactivity and explosives from the areas. All radioactive pollutants found on these territories have been moved to an appropriate disposal site.

Environmental risk prevention programmes are being developed, based on the principle that the first action must be to remove pollution sources in order to prevent further pollution. Soil polluted with petroleum products is considered to be a secondary source of pollution. According to existing requirements, industrial activity in areas polluted with petroleum products is permissible only when the concentration of petroleum products in the ground is less than 2000 mg/kg.

Ten military sites have been investigated in detail to date. Based on these data, new projects related to ecological health and reuse of other areas will be prepared. For instance, the results hey suggest what preventative works must be carried out immediately in other similar locations. Such areas have been selected in various types of Lithuanian areas so that they reflect different pollution sources and the typical landscape damages that are seen. Monitoring, as a major environmental risk preventative measures, is obligatory for both the surroundings of cleanup sites and source areas that have been isolated and where the spread of pollution has been prevented. In some cases, monitoring helps track the cleanup process and evaluate the level of cleanup achieved, but it is not obligatory after cleanup has been successfully completed. In other cases, monitoring can last 10-20 years and follows the process of natural attenuation and an assessment of the risk of the spread of pollution and how that spread might be prevented.

The majority of military sites that are located in urban zones are used for living or industrial operations. Ecological health of the area requires agreement on its functions. In such areas we have determined that the first action should be to remove all sources of pollution from the ground's surface, and sensitive habitats. These areas could capture streams of pollutants.

The Lithuanian Ministry of Defence has taken over about one third of Lithuania's former Soviet military territories, but [it does not have funding to take care of all its territories properly. The Ministry of Defence, in cooperation with the Ministry of Environment and environmentalists from other institutions, have listed twenty priority military sites that need to be restored as the priority areas. A Joint Committee is currently being created by the Ministry of Environment and the Ministry of Defence. Its function will be to organise environmental risk preventative activity at working military sites that are currently in use by Lithuanian soldiers. The Committee will prepare a methodology for resolving environmental protection problems at active military sites in order to prevent the creation of new sources of pollution.

Scenarios of ecological protection and site redevelopment are created according to the restoration and optimization of typical military sites. According to the plan of optimization and renovation, the first problems likely to be encountered is

localization of contamination hot spots and the funding of funding for site restoration. At military sites that exist in uninhabited locations, the first priority is to restore the sites according with their desired future use and the goal of preserving ecological stability at the location.

6. Environmental Management and Risk Prevention in Military Sites

In according with the military's environmental management strategy, which has been accepted by the Lithuanian Ministries of Nature and Defence, every operational military site in Lithuania must have an environmental management plan that has been approved by the Government. The pilot environmental management plan is titled "Environmental Base Management Plan for Central Training Areas of the Lithuanian Armed Forces." The project was initiated following an information seminar, organized by the Swedish Armed Forces and the U.S. Environmental Protection Agency, on the base management plan for the Adazi Central Training Area (Latvia). Pursuant to the Joint Order No. 83/51 of the Minister of National Defence and the Minister of the Environment, a working group was established for drawing up the Pabradė Base Management Plan. It consisted of specialists from the Lithuanian Ministries of National Defense and Environment, as well as from the faculty of the Vilnius Gediminas Technical University (Kaunas, Lithuania).

This working group then launched the development of the Environmental Base Management Plan of the Pabradė Training Area of the Lithuanian Armed Forces, which could help the governing body of the Pabradė Training Area to solve environmental issues following set priorities and proposed measures and to ensure necessary environmental protection in the Pabradė Training Area. The Pabradė plan will serve as a model for management plans for environmental protection for other units of the Lithuanian Armed Forces.

The project is aimed at ensuring due protection of the environment, as well as natural resources upon planning military training in the territory of the Pabradė Training Area. Proper care for the environment will ensure long-term use of the territory for military training and the protection of human health and biological diversity. A very important part of this project is related to the problems of optimization interactions between necessary military activity and environmental stability and restoration. In order to be able to set priorities in respect to environmental activities at the Pabradė Training Area, a risk assessment has been carried out using the Scandinavian model of integrated facility's environmental risk assessment. The assessment is based on the following factors:

- The risks of spreading pollution (can range from low to very high)
- The pollution level or the extent of activities having a negative impact on the environment (low – very high)
- Environmental sensitivity as regards pollution or activities (low – very high)
- The danger of the pollution or activity to the environment (low – very high)

Having assessed the aforementioned factors, the risk assessment is divided into four classes:

Class 1 – measures to be taken to reduce negative environmental impacts or treatment of the polluted areas; must be performed within the next 1-2 years.
Class 2 – planning and more detailed analysis that must be underway in 2-3 years.
Class 3 –pollution and/or dangerous activities are taking place, but the situation is not very urgent, so measures may be planned to occur in 3-5 years.
Class 4 – Pollution or dangerous activities are taking place, but it is enough to note their occurrence them, and no special measures should be taken. The results of this environmental risk assessment are presented in the Appendix 1.

7. Conclusions

The negative effects of military activities on the environment have been determined mainly by the kind of activity, as well as by the peculiarities of the local natural environment. Information regarding Lithuania is presented as a case study. Intensive military activities have done great harm to the environment. Nevertheless, military grounds, due to their closed character, have often formed conditions favourable to the survival of natural ecosystems. Since we seeking protect of the environment and natural resources, as well as to lessen environmental risk due to future planned military activities, all active military sites must have an approved environmental management plan. Environmental risk prevention and management of military sites in Lithuania demands effective, expensive and urgent maintenance and restoration measures.

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Appendix 1. Environmental Risk Assessment in the Central training Area of the Lithuanian Armed Forces

INFRASTRUCTURE SUPPLY SYSTEM

	Water supply system	**Waste water system**	**Surface drainage**	**Fuel supply system**	**Heating system**	**Electricity supply system**	**Solid waste management**	**Premises and surfaces**
Risk of spreading	low	high	medium	high	low	low	medium	low
Pollution level/ activity extent	low	medium	medium	medium	low	low	medium	low
Sensitivity of the area	high	high	high	high	low	low	high	medium
Danger to environment	low	high	medium	very high	low	low	medium	medium
Long-term effects	low	high	medium	very high	low	low	medium	low
Class	**4**	**2**	**3**	**1**	**4**	**4**	**3**	**4**

MILITARY TRAINING

	Shooting/ blasting	**Driving vehicles/ training to use military vehicles**	**Training military units**	**Incidents/ Accidents**
Risk of spreading	medium	low	low	medium
Pollution level/ activity extent	medium-high	low-medium	low	low
Sensitivity of the area	high	high	medium	medium
Danger to environment	high	low-medium	low	low-high
Long-term effects	high	low-medium	low	low-high
Class	**3**	**4**	**4**	**3**

CIVIL ACTIVITIES

	Wash-house	**Workshops**	**Medical station**	**Cleaning of premises**	**Canteen**
Risk of spreading	high	medium	low	low	low
Pollution level/ activity extent	medium	medium	low	low	low
Sensitivity of the area	high	high	low	low	medium
Danger to environment	medium	medium	low	medium	medium
Long-term effects	medium	medium	low	low	medium
Class	**2**	**3**	**4**	**4**	**4**

ENVIRONMENTAL RISK MANAGEMENT ISSUES IN ROMANIA - ECONOMIC INFORMATION POLICY IN A TRANSITION PERIOD

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Introduction

After 1989, Central Europe has been confronted with a special situation euphemistically called "period of transition." We could say that there are a few regions in the world that have undergone such profound and numerous changes in such short period of time. Developing national environmental policies in countries with economies in transition means defining goals and priorities and selecting projects to be funded. These decisions are eventually associated with the allocation of scarce resources and raise questions about the fairness of distribution, the insufficient perception by decision-makers of risk, and lack of responses at the policy implementation, while orientated primarily toward short-term economic information. The "forgetfulness," the sustainability principle in the information policy, needs to identify some potential risks: in most cases, multiple principles and criteria of fairness underlying the selection of goals and interventions in Romania's long-term environmental plan.

A project aimed at developing the long-term Environmental Risk Management Plan for Romania was initiated and sponsored by the Romanian Academy of Science and the National Commission of Economic Restructuring. Results and conclusions were recently proposed to the Romanian Ministry of European Integration which aimed to grant Romanian Legislation to European Union agreements in the field of applicable European Directives. The plan includes:

(i) defining the most important target variables, assessing their current values and proposing desirable values (goals) for the years 2000 to 2010
(ii) defining the most significant factors influencing goals
(iii) generating possible environmental policy interventions, doing the analysis of the impacts of the most cost-effective set of interventions, in terms of potential impacts, costs, and feasibility
(iv) analyzing the impacts of the most cost-effective set of interventions contingent upon various socioeconomic scenarios
(v) developing recommendations for national environmental policy

The environmental plan included the project "Risk Assessment of Complex Technological Systems, Development of Strategies and Policies in Environmental Management." The goals identified by the study were related to air and water quality, the state of soil and biota, waste management, and sewage treatment.

I. Linkov and A. Bakr Ramadan (eds.),
Comparative Risk Assessment and Environmental Decision Making, 413–422.

In bridging the gap between Eastern and Western countries in the approach of risk assessment and environmental impact analysis, we notice also failure to compare the environmental and economic impact of different macro- and microeconomic policies in such area as trade, investment, and the use of economic instruments.

The author proposes the very general model of environmental risk management and shows that for the problem of trans-boundary risk management, a mutually acceptable cooperative solution can be found. A solution of this kind may serve as a foundation for environmental parity.

Project Description

Industrial development is essential to improving the standard of living in all countries. In any given region, old and new plants, processes, and technologies must coexist. Dynamic technological penetration and substitution processes are taking place, and this trend will remain. Managing the hazards of modern technological systems has become a key activity in highly industrialized countries. Decision-makers are[1] often confronted with complex issues concerning economic and social development, industrialization and associated infrastructure needs as well as population and land-use planning. Such issues must be addressed so that public health will not be disrupted or substantially degraded.

While hazard managers and risk assessors have successfully identified hazards and reduced overall risk exposure, economic growth and technological development have led to a new risk situation characterized by:

- an increasing number and variety of hazards
- hazards giving rise to a broad range of partial and temporal risks
- public dissatisfaction with hazards managers and hazards owners

Due to the increasing complexity of technological systems and the higher geographical density of point hazard sources, new methodologies and a novel approach to these problems challenge risk managers and regional planners. Risks from these new complex technological systems are inherently different from those addressed by risk managers of the '60s and '70s. Recent awareness of environmental problems by a large public has led to worldwide dissatisfaction and the formation of all kinds of pressure groups that exert a strong political influence and are quite often not ready to accept any compromise. As a result of this approach, interesting and vital research projects were stopped, plans have been buried, and decisions delayed for many years (i.e., nuclear waste repositories). Nowadays in former communist regions, it becomes increasingly difficult to site new plant facilities perceived as risky or undesirable by the local population. Projects of national interest cannot be launched or even achieved, once started. Distrust has become so problematic that, at times, risk managers are no longer considered impartial or reliable sources of information regarding risk protection.

It is felt that existing hazards management techniques need to be supplemented with concepts and methods applicable at a regional level. Integrated regional risk

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assessment and safety/hazard management represents a coordinated strategy for risk reduction and safety/hazard management in a spatially-defined region across a broad range of hazard sources (during normal operation and accidental situations) that includes synergistic effects.

In view of the above, the Safety Culture, Environmental Awareness, and Emergency Culture for Romania's Risk Management (Andreas, 1999) project was Romania's Academy of Science and launched by the Romanian Association for Science and Progress in Bucharest.

Safety culture, public participation, and risk communication are relevant to the overall landscape of regional risk assessment. Emergency culture, preparedness, and planning are integral to regional safety management. A series of basic questions was asked, the answers to which should have included the following main issues:

- definition of integrated area risk assessment and safety management
- how to define a region/area for study
- type of activities and targets at risk
- objectives and scope
- need for risk impact indicators
- need for a comprehensive methodology

Regional risk assessment and safety management seems to be a medium that helps integrate people, issues, and decisions in area risk assessment.

Assessing the risks of a region implies the use of a complex methodology dealing with risks to health and the environment, normal operation and accidental situations, and a large variety of industries, impacts, regulations, and players. A task-oriented approach allows a systematic analysis of the problem of regional risk assessment, flexibility and efficiency in implementation and initiatives, and ad hoc modeling and simulation.

Integration of risks cannot be done through a single risk indicator. Integrated regional risk assessment should be a process whereby decision-making techniques (ranging from simple brainstorming [the Delphi method] to multi-criteria decision analysis, and decision and knowledge support systems) play an important role. Various techniques must complement expert judgment, public participation, and risk communication. One must balance hard approaches (models, calculations) versus soft approaches (acceptability) in regional risk analysis. Comparative risk assessment should play an important role in risk integration.

New information technology (e.g., artificial intelligence such as expert systems, fuzzy logic, and neural networks; multimedia; virtual reality; geographic information systems; specialized relational databases; and computer graphics) will play a significant role in the future of regional risk assessment and safety management practice.

Methodological key issues (Gheorghe, Andreas, 1994, 2) to ensure a credible comparison among different technologies or technological systems in a given region can be summarized as follows:

a) The delineation of consistent and comparable boundaries for different technologies or chains of technologies. The definition of principles and delineation of "reference technological chain" for risk comparison purposes is a priority issue in this regard.

b) Methods and dose-effect relationship for the risk assessment of the health effects of different technological systems (e.g., chemical plants, electricity generation systems, biotechnological installations) including, in particular, the derivation of consistent indicators of health risks for comparative assessment.
c) Methods and relationships for assessment of the impacts of different technologies on the natural environment and associated ecosystems and, particularly, the derivation and application of indicators of environmental risks for comparative assessment.
d) How to deal, in the comparative risk assessment process, with the time and technological dependencies of risk estimation, given variations in technologies and technological development over time. Can the principles of "risk discounting" be used to compare existing risks by postulating and accounting for possible future technological development? How can future technological development be accounted for in the comparative assessment process?
e) How to deal in the comparative risk assessment process with the issue of "uncertainties." Such uncertainties exist in the type, nature and extent of various impacts and in the variable nature of some impacts over time. Uncertainties also exist in the estimation (both qualitatively and quantitatively) of the various risks, both in terms of consequences and probabilities.
f) Comparative risk assessment for severe accidents in different technological systems. The main issues here relate to the tools and criteria for comparison; methods for presenting the results; and the comprehensiveness of the comparative results, particularly in relation to environmental impacts from severe accidents.
g) The integration of different elements of risks, including whether one overall indicator of risk is possible or feasible; the applicability of comparative risk assessment to regional safety management studies; and the weighting of various dimensions of different risks in the comparative risk assessment process.
h) Methods and tools for the presentation of the results of comparative risk assessment.

In the broadest terms, regional hazard management can be defined as a coordinated strategy for risk reduction and hazard management in a spatially-defined region across a broad range of hazard sources that includes synergistic effects. Regional hazard management is also a multi-disciplinary process. While engineers and computer scientists will continue to play central roles, particularly at the risk assessment stage, other disciplines make essential contributions as well. Social scientists can make important contributions toward understanding how the practice of regional hazard management can meet society's needs and expectations. Finally, a regional approach to hazard management can be more easily brought into other kinds of regional planning functions such as waste disposal, infrastructure, zoning, building and development, and emergency preparedness. As such it requires the expertise of planners.

The environmental plan included the project: Risk Assessment of Complex Technological Systems, Development of Strategies and Policies in Environmental Management, for the Region of Ploiesti, Romania (Andreas, 1998, 3). The region of Ploiesti is an area of 3,000 km^2 and 1,700,000 inhabitants (working mostly in

agriculture with 30% in industry) in the southern part of Romania where almost all oil reserves are concentrated. The area includes oil extraction facilities; three medium-sized refineries and oil processing facilities; and a railway transportation node of Romania with various rail- equipment-repair facilities. The city of PLOIESTI, with 500,000 inhabitants, is situated 60 km from Bucharest, at the southern part of a hilly region with the Carpathian Mountains to the north and the Danube Plain to the south.

The study goals were to:

- identify existing hazards of technological systems within the region
- identify some of the criteria needed to evaluate the risks from these hazards
- assess and evaluate the risks posed by these hazards
- coordinate hazard management activities with land use planning
- suggest regulatory mechanisms to reduce risks and eliminate hazards
- design emergency response plans

We limited discussion to those kinds of hazards associated with the hardwired part of the technological system. These can be divided into two types:

- Those that realize harm or damage through accidents and that can be attributed to technological failures, human error, and natural interventions
- Those that realize damage or harm through routine (continuous) emissions to air, water, and land.

In both instances, we were concerned with the risks these hazards pose for the region's people and environment. Each hazard source, accompanied by a risk profile, was studied illustrating the quality of the risk and the magnitude of possible damage. This step included a discussion of exposure modeling, fault and event tree analysis, and other risk assessment methodologies. As examples, we have taken three topical areas of research in the study:

- risk assessment of landfills
- risk assessment of hazardous goods transport
- dispersion modeling of chemical clouds (Fig.1-Description of the Toxic Effect Model and associated submodels)

Risk evaluation is an inherently political process. The challenge is to integrate technical knowledge; knowledge from the social sciences, decision sciences, safety culture, and political science, and the local knowledge and preferences of citizens, politicians, and stakeholders groups in the region.

Translating risk assessments and risk evaluations into a feasible plan for hazard management is one of the most difficult steps of the entire process. Risk evaluations should be reasonable, taking into consideration the politically and economically feasible. The overall goal is to produce a basis for making rational and accountable decisions about an integrated and coordinated hazard management plan for the region.

The resulting plan should deal with all major hazard sources in the region over the entire production and operations cycle, considering all impacts (technical, social, health, and economic). Most of those who worked on developing these studies are scientists and engineers who have worked in the field of risk assessment and environment for some time, so it is natural that certain prejudices shows through (for instance, a general belief in the appropriateness of estimating the risk of low doses by

extrapolating a linear dose response curve from data on high level doses in non-human species).

The principles of setting boundaries for comparative risk assessment of different technologies for regional safety management are:

- Setting boundaries must be adapted to the purpose of the assessment
- Boundaries should take into account:
    - time (horizon)
    - space (workers and public exposure, environment)
    - phases of the systems under investigation
    - impacts to be considered
    - material balance consideration in terms of risk
    - per net production (e.g., energy) output
    - acceptable threshold criteria
- At a regional level, the setting of boundaries will greatly benefit from a structured public scoping/participation process, so all concerned can be involved in boundary setting. Such a process greatly enhances the credibility and acceptance of the study.

Usually in a region where complex risk assessment studies are made, one must consider that a large number of compounds are released into the atmosphere and water and disposed of on land with associated environmental impacts (risks), immediate or long-term. Implicit "environmental impacts" are often a direct or indirect "inference" of "health impacts." Table 1 and Table 2 show a generalized environmental transfer model, outlining the various essential components of health and environmental risk estimations in large industrial regions.

Table 1 Categories of risk usually adopted to assess and compare the health impacts of different pollutant sources

Health Risk			
Source	**People at Risk**	**Exposure**	**Effects**
Routine or accidents	Workers and Public	Short or Medium and Long-term	Fatal and Non-fatal Immediate/delayed respectively Long-term/delayed

Table 2 Categories of risk usually adopted to assess and compare the environmental impacts of different pollutant sources

Environmental risk		
Source	**Effects**	
	Duration	**Extent**
Routine or Accidents	Short or Medium and Long-term	Local, regional and global

Figure 2 shows a linear pollutant pathway model indicating the amount reaching the receptor (target at risk) as the function of the amount emitted, altered by dilution and removal and enhanced by environmental accumulation factors.

As shown, distance (space) and rate of movement (time) are critical parameters to an environmental impact assessment study. The complex interdependencies among time, space, and feedback mechanisms (degree of resilience of a given environment to external factors) are not fully known; very often, it is difficult to normalize them on a common scale for comparison.

In the comparative risk assessment of different technologies one has to "compare risks" which may be different in a subjective way from impacts, effects, emissions, etc. Two major limitations have been considered when dealing with the assessment of environmental impacts of different technologies or technological chains, namely:

- the effects are not always susceptible to quantification
- no general agreement exists on what should be quantified

Methods of relevance for making comparisons of environmental impacts in integrated regional risk assessment and safety management studies are:

- ranked matrix environmental assessments
- emission values and ambient quality indices
- critical loads and critical levels

Critical Levels: concentrations of pollutants in the atmosphere above which direct adverse effects on receptors such as the ecosystem, may occur, according to present knowledge. Critical Loads: quantitative estimates of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge.

Quantitative and qualitative indicators used for environmental impact indicators of technologies were: site specific, society specific, and value judgment dependent. Synergistic effects are to be taken into consideration when dealing with comparative impact assessment. Environmental performance measurements already accepted by international organization and practice and used in comparative analysis are:

- environmental performance indicators (e.g., river quality, air quality, soil quality)
- environmental goals (critical loads, sustainability index)
- environmental emissions (SOx, NOx,CO2, etc.)

The computer simulation tool, the Inter-CLAIR-Environmental Decision Support System, integrates basic knowledge from environmental engineering, mathematics, natural sciences and economical modeling; and simulates specific processes for ecological, economical, and risk assessment in regions with dimensions up to 400x400 Km.

Other applications included: database of regional stationary pollution sources, database of technological measures (new clean technologies, optional filters, fuels, liquidation of emission source, etc.); and data on industrial and source-specific impacts on the atmosphere, ecological and economical effectiveness analyses of overall industrial innovations, optimized investment allocations for air quality management, air pollution optimal reduction within specified expenditure constraints, the effective set of

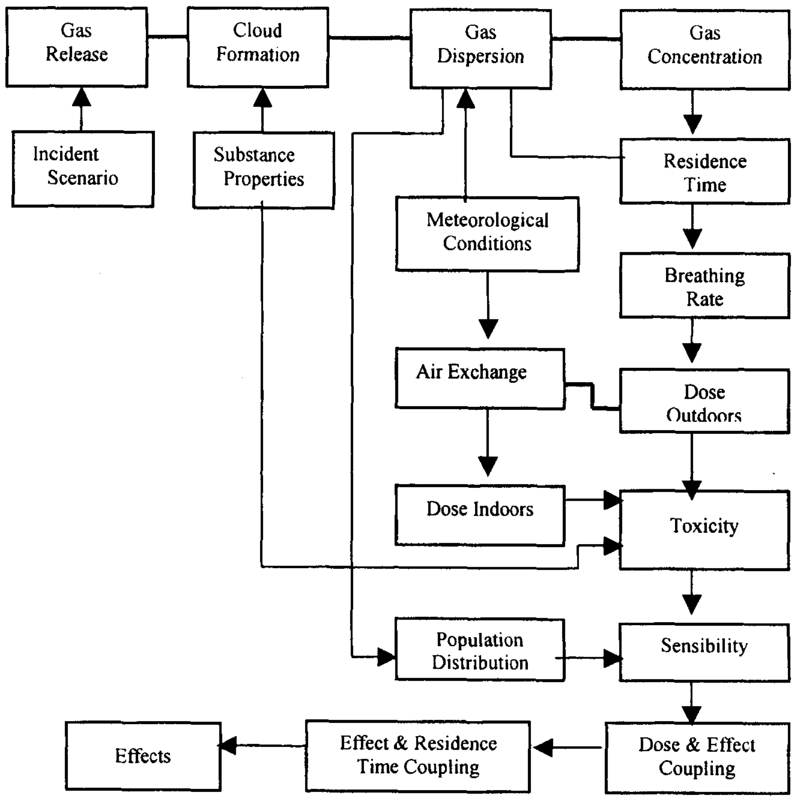

Figure 1. Description of the Toxic Effect Model (TEM) and associated sub-modules

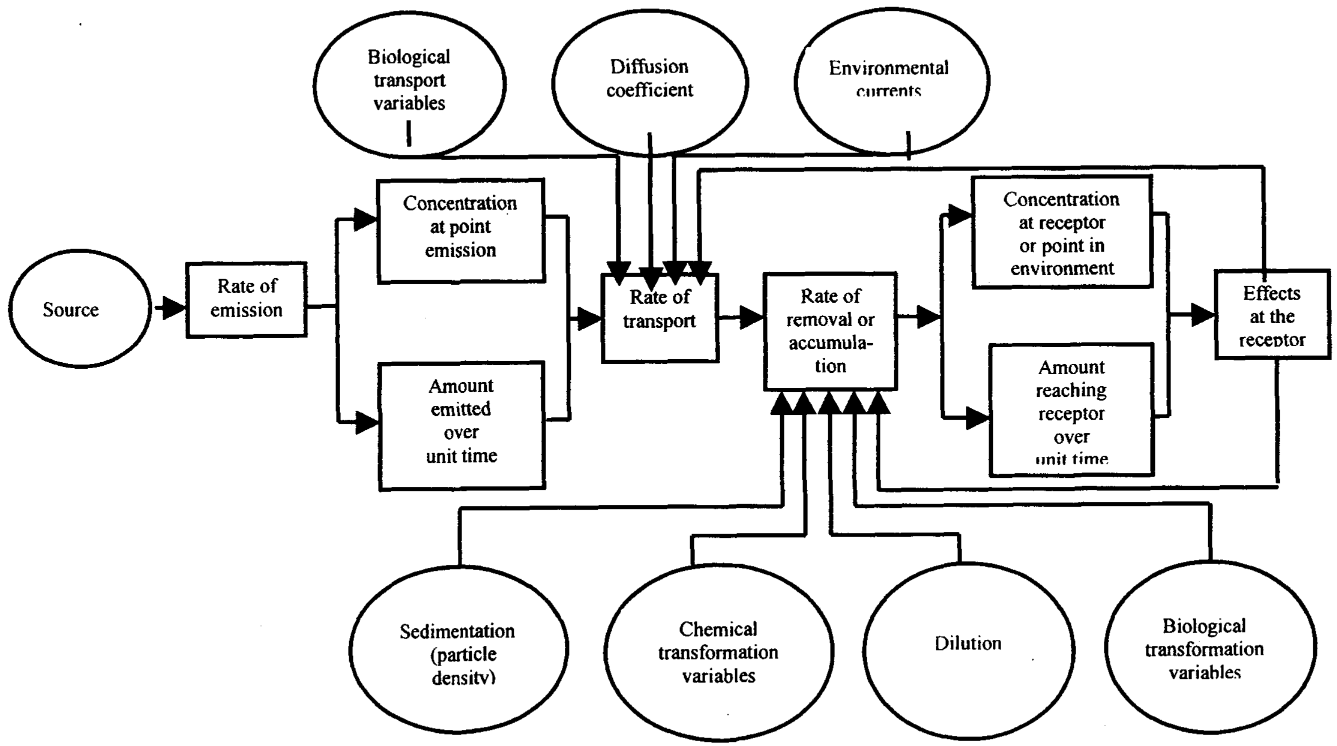

Figure 2. Pollutant pathway model used for assessing environmental impacts from emissions

technological measures that could be applied to any emission source, and critical load estimation for various environments and receptors.

Optimal simulation solutions abatement can be calculated simultaneously in Inter-CLAIR for any given set of pollutants (SOx, NOx). Required data to run the system includes :

- emissions sources data
- technological measures data
- meteorological data for the region (averaged for 10 years)

Resulting data and maps available as potential decision-makers include :

- maps of emission distribution for a given number of pollutants
- maps of economical and environmental damage distribution
- map of pollutant concentration distribution in the region
- maps of wet and dry deposition for sulfates and nitrates
- sulfates and nitrates transportation diagrams of the area considered
- maps evaluating danger (due to acid rain) to coniferous forests
- graphs of optimal strategy as a function of expenditures for atmospheric pollution reduction from each selected criteria (emissions, concentrations, depositions, transportations out of the region) and a combination thereof

In bridging the gap between Western and Eastern European countries in their approaches to risk assessment and environmental impact analysis, we notice also the failure to compare the environmental and economic impacts of different macro- and microeconomic policies in such areas as trade, investment, and the use of economic instruments. In dealing with the problem of trans-boundary risk management, a mutually acceptable cooperative solution may serve as a foundation for environmental parity.

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A BRIEF HISTORY OF RISK ASSESSMENT AND MANAGEMENT AFTER THE SEVESO ACCIDENT

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Abstract

After the diffusion from the Icmesa plant of the toxic cloud on 10^{th} July 1976, there was a mobilization of the Public Authorities and the Italian Scientific organizations, especially the ones in Lombardy. Among the committees that were created during the emergency, one can mention the followings: for Reclaiming, for the Chemical (and Statistical) Analysis, for Research, for Epidemiology, Toxicology etc., etc.

The task of fixing acceptable limits, that is, of establishing the first "Risk Assessment" for TCDD in the soil in the Brianza area of Seveso, was carried out by Technicians and Scientists of several committees, with the constant supervision of the Regional Authorities and the cooperation of the Istituto Superiore di Sanità – I.S.S. – of Rome (National Health Institute).

The basic criteria were simple:

a) NOEL (no observed effects level) on the most sensitive animal, that is the guinea pig:
1 ng/Kg-body weight per day;
b) Uncertainty factor: 1000;
c) Inhabitants most exposed to risk (of ingestion of contaminated soil): children;
d) Limit dose found: 1 pg/kg-body weight per day (equal to approx. 0,14 gr. of soil ingested by a child weighing 20 kg, or 0,50 gr. for an adult weighing 70kg.);

From the above criteria, the acceptable amounts on the various matrixes were assessed: land, inside of housing, equipment and others as listed in the text of the Regional Law for Seveso. The defining of the risk areas was thus carried out taking into account only the contaminations found in the soil and measured on couples of samples (carrots), extracted by means of steel cylinders having a diam. of 7 cm and penetrating into the soil for about 7 cm.

From then onwards and for all operations relative to the reclaiming and re-checking, reference was made to the unit $\mu g/m^2$, always intending a square surface of soil having 1 meter long sides and 7 cm deep. The conversion from $\mu g/m^2$ to the most commonly accepted ppt (weight of contaminating agent by weight of soil) is obtained to a good approximation by multiplying the number of $\mu g/m^2$ by factor 8 and therefore:

I. Linkov and A. Bakr Ramadan (eds.),
Comparative Risk Assessment and Environmental Decision Making, 423–429.

Agricultural land - limit by law	0.75 $\mu g/m^2$	= 6 ppt
Non agricultural land – limit by law	5.00 $\mu g/m^2$	= 40 ppt
Limit for evacuating population	50.00 $\mu g/m^2$	=400 ppt

It is surprising to note the difference between the Regional limit in Lombardy for evacuation especially considering all the economic and social consequences deriving from such action, and the limit "of concern" – but not yet of intervention – indicated by the Sanitary Control Authority of the United States (Center for Disease Control), which is 1 ppb = 1000 ppt, that is 2,5 times higher than that adopted in Seveso: a different risk assessment of the presence of TCDD in the environment. The roads of Times Beach (MO) which had been sprayed with contaminated oils, were covered with a layer of asphalt to avoid contact with a contaminations of 300 ppb. Times Beach was (gradually) evacuated after the river had flooded the area and some analyses had shown the intensity of TCDD contamination.

In 1982 and in 1984/85, during the Seveso reclaiming, the "B" zone was also re-checked – this area has a contamination between 5 and 50 $\mu g/m^2$ – and confirmation of the unequal trend of the contamination was obtained, as tough the cloud had "bounced" on the land various times, leaving a toxic print less and less marked every time. Among such contaminated spots, the cloud left lower levels of TCDD (see the tridimensional representation). Therefore it was necessary to carry out a more detailed "risk assessment" than the one initially carried out 9 years before during the emergency period, taking into account the population at risk (approx. 6000 people) residing in the "B" zone.

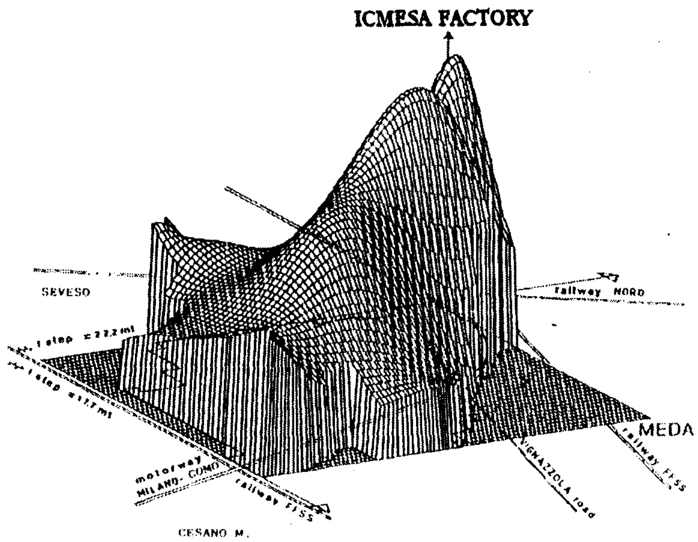
In the document elaborated by the participants to the sub-committee of the "Cimmino" Committee for Seveso, organized by the Special office of the Lombardy Region and by I.S.S., in the meeting held in Milan on 26-30/03/84, it was shown that a daily ingestion of TCDD of 0.280 pg/Kg of body weight, increases the risk of cancer by 1/100.000, if we consider for the calculation, the effects observed on the liver of female rats. In fact the liver of female rats were considered to be the most sensitive organ to the toxic compound.

Moreover, it was assumed that the correlation between the risk of cancer and the dose of TCDD is linear. So, 0.028 pg/Kg- body weight per day reduce the risk to $1*10^{-6}$, while 2.8 pg/Kg increase it to $1*10^{-4}$ and so on.

The contamination of Zone "B" was schematically assumed as follows: for 80% of its extension: 1.5 $\mu g/m^2$ = 12 ppt of TCDD in the soil. The remaining 20% with 10 times higher contamination. The means of exposure considered are:

1.0 Ingestion;
2.0 Contact with the soil (cutaneous absorption);
3.0 Consumption of epigean and hypogean vegetables grown in the Zone "B" gardens.
4.0 Consumption of zootechnical products (chickens and rabbits) from that area.

On the basis of a series of hypotheses, which were only partially verified, the following table may be obtained, assuming that total risk is given by the sum of the single exposure risks.



Tridimensional map of the TCDD contamination according to mathematical model

Soil at 1.5 $\mu g/m^2$ = 12 ppt. A person weighing 70 kg. Length of permanence in the area: 70 years. Family having a vegetable garden and courtyard animals.

Exposure routes	Q ingested pg/Kg-body weight per day	Additional cancer risk per million exposed	Comments
Soil ingestion	0,007	0,25	Equivalent to 3/10.000. This dose is associated to a cancer risk of 1/3330
Vegetables ingestion	5,700	200,00	
Animals ingestion	2,400	100,00	
Total exposure	8,107	300,25	

On the basis of the above results the sub-committee of the Cimmino Committee for Seveso recommended a more detailed analysis of the various parameters, especially the ones that had not been thoroughly verified such as the relationship between contaminated soil and vegetables, to check, by means of a survey on the territory, the feeding habits of the population (including the consumption of eggs and milk not considered in the above table) in order to reach a more reliable evaluation. Such detailed analysis was to be carried out also by creating experimental cultivated

plots and animal farms in Zone B. At the same time the dilution/degradation was to be helped through agronomical interventions.

The U.S. EPA, in one of its publications, has defined a method of risk assessment for soil contaminated by TCDD (EPA-600/8-84-031 nov 84). In this survey five means of intake are considered, that is: dust inhalation, cutaneous absorption, ingestion of soil and ingestion of food such as veal meat and dairy products, fish consumption. The interest of this publication is in the discussion of the coefficients for the evaluation of the amounts absorbed by every route. For a fast and approximate estimate of the quantities ingested and therefore of the cancer risk, the use of nomograms is suggested. More thorough procedures enable to reach more accurate risk assessments for TCDD exposure.

For Zone "B", using the simpler method of nomograms, the following values are obtained, expressed in number of additional tumours every million inhabitants exposed. The soil is considered - as in the previous case to be 1, 5 $\mu g/m^2$ = 12 ppt.

Inhaling	:	dust in air (inhalable): $2mg/m^3$ N° tumours 0,4 / 1 million inhabitants
Cutaneous absorption	:	N° of tumours $1*10^{-5}$ / million inhabitants Neglectable
Soil ingestion	:	N° of tumours 0,1 / 1 million inhabitants. This is the only hypothesis of risk evaluated in 1976
Meat and dairy products	:	N° of tumours 100 / 1 million inhabitants
Fish consumption	:	In Zone "B" there does not exist any river where one may fish and so the risk of contaminated fish consumption is <u>zero.</u>

By adding all the single risks we obtain a total risk as follows:

Total number of tumours = 0,4 + 0.00001+0.1+100+0,0 = 100,5/million

Since the residing population is slightly less than 6.000 units (and only partly exposed to the risk of feeding with products grown in zone "B") this value has given someone the impression that the percentage of the single additional tumour (on 10.000 inhabitants) could not be in any case statistically quantifiable!

However, just in view of this additional-risk the Special Office of the Lombardy Region decided, with the support of the Hoffmann-La Roche Concern, which acted in a responsible way, to proceed to a series of operations for diluting and even replacing the contaminated soil, in the vegetable gardens and agricultural areas, with the aim of further reducing the risk deriving from feeding with vegetables and farm animals bred in that area.

We owe Prof. Schlatter of the Swiss Federal Institute of Toxicology, a clear presentation of "Risk Assessment" by 2,3,7,8 TCDD at "Dioxin 85" held at the Bayreuth University in the fall of that year.

Having examined the ADI (Acceptable Daily Intake), set in 1-10 pg/Kg-body weight per day, Prof. Schlatter lists the results of the latest researches of his Institute:

a) The amount accumulated in the human body is known;
b) The isomers (12) having toxic relevance are known;
c) The main origin of TCDD in man is known: the intake of animal fats;
d) The half-life period of TCDD in the human body is known, and this thanks to the experiments carried out on himself by Dr. Poiger, assistant to Prof. Schlatter: 5-6 years, 80 times longer than that in laboratory animals.

The population of the industrial areas, where the fumes and ashes of the municipal waste incinerators fall, assume 16 pg/Kg-body weight per day of toxic equivalent dioxins, i.e. of the 12 isomers having toxicity equalized to that of 2,3,7,8 TCDD. In view of the above, the proposed limit for ADI is largely exceeded and it is necessary to examine the problems, starting from the effects on the health of the populations exposed to low level contamination for a long period of time (chronic exposure).

The only symptoms encountered in Seveso are those of Chloracne, which appear only in doses of many ng/Kg-body weight per day.

In view of the above, Prof. Schlatter concludes that the exposure of the population on industrial areas to this class of toxic compounds (as evaluated on the basis of the amount present in the adipose tissues) is at least 100 times lower than the critical limit (at which Chloracne appears).

Even though ADI has been widely exceeded by reality, concludes Prof. Schlatter, the final aim must not be forgotten. The intake of dioxins must be reduced below the fixed limit of 1-10 pg/Kg-body weight per day and there is still a long way to go to reach such results.

Final Consideration

In 1976 the relative hazardousness of TCDD was practically unknown.

The previous accidents in Philips-Duphar (NL) and at BASF (D), where the diffusion of dioxin was confined within the plants, and only one worker of the reclamation team in Ludwigshaven plant undressed protective mask and died apparently affected by TCDD. The precaution taken in the Seveso area was very strict and carefully followed by the population. 735 person were evacuated and only part of this group came back in the A6-A7 zone when the homes and gardens were totally reclaimed and checked. For nearly 8 years the fields of the B-Zone were not harvested and the vegetables were mixed into the soil by ploughing; in addition no animals or vegetables grown in the area were allowed for human consumption. Thanks to all these precautions, no acute pathology were found with the exception of the Chloracne for about 85 individuals below 14 years of age.

The long-term epidemiologic study on possible carcinogenic effects was negative in general: however some types of tumours (of digestive tubes, of respiratory apparatus and of linfatic and emopoietic tissues) slightly increased. This result was achieved after a follow-up, which lasted twenty years; as already stated the Seveso population was mostly exposed at the beginning of the accident while, when all the precautions were in force, the exposure was reduced down to nearly zero. The ADI originally placed at 1pg was lately increased at 10 pg/Kg-body weight per day by the US C.D.C.. In effect the limit enforced was much higher then the background contamination in Lombardy region, as proved by prof. Schlatter and Poiger in their studies.

While reducing the amount of toxic compounds in the environment, any improvement of the risk assessment methods is necessary and useful in order to control in a responsible way the risk.

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AVERAGE TCDD INTAKES ESTIMATED FOR SELECTED EXPOSURE GROUPS OF RESIDENTS IN ZONE B
(TCDD LEVEL IN SOIL = 12 PPT) (pg/Kg - body weight per day)

EXPOSURE GROUPS	VEGETABLES			COURT-YARD ANIMALS			OTHER ANIMALS		TOTAL*
	HYPOGEOUS	EPIGEOUS	FRUIT	POULTRY	EGGS	RABBIT	PIG	BEEF AND SHEEP	
1. PEOPLE NOT CONSUMING LOCALLY PRODUCED FOOD	---	---	---	---	---	---	---	---	0.035
2. PEOPLE CONSUMING LOCALLY PRODUCED PLANT FOOD	0.8	1.3	1.4	---	---	---	---	---	3.5
3. PEOPLE CONSUMING LOCALLY PRODUCED PLANT AND ANIMAL FOOD (LITTLE COURT YARD ANIMALS)	0.8	1.3	1.4	2.3	1.7	2.7	---	---	10.2
4. PEOPLE CONSUMING LOCALLY PRODUCED PLANT AND ANIMAL FOOD (ALSO LARGE ANIMALS)	0.8	1.3	1.4	2.3	1.7	2.7	12.2	10.9**	33.3

* IT DOES NOT TAKE INTO THE ENVIROMENTAL DEGRADATION OF TCDD

** THE SHEEP MEAT CONSUMPTION IS ASSUMED TO REPRESENT NOT MORE THAN 6% IN THIS CLASS

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